National Oceanic and Atmospheric Administration

Revised Final Report

SOUTHERN CALIFORNIA BIGHT DAMAGE ASSESSMENT FOOD WEB/PATHWAYS STUDY

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HydroQual, Inc.
Environmental Engineers and Scientists

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SECTION 1

SUMMARY AND CONCLUSIONS

1.1 PURPOSE

Evidence that biota in the vicinity of the Palos Verdes Shelf have elevated concentrations of DDT and its metabolites (approximately 90 percent p,p'DDE) and PCBs has existed since about 1970. Recent reports summarizing contamination in the Southern California Bight include Risebrough (1987), Mearns et al. (1991), Pollock et al. (1991), Southern California Coastal Water Research Project (1992) and Los Angeles County Sanitation District (1995). The Whites Point wastewater treatment plant outfall is generally considered to have been the source of most of the DDT contamination and much of the PCB contamination in the Southern California Bight, based primarily on spatial and temporal correlations observed in contaminant data for sediments, mussels and fish. Examples of such correlations are shown on Figures 1-1 and 1-2[1]. The spatial and temporal patterns are robust, in that data collected in a variety of studies show similar profiles. Although contaminant loads from the outfall declined precipitously in the early 1970's, DDT and PCB contamination in the fish, birds and sea lions has persisted since that time. Sediments contaminated by the outfall discharges are the probable source of contamination to the biota since the mid 1970's.

For fish species such as the Dover sole (Microstomus pacificus) and the white croaker (Genyonemus lineatus), a direct pathway of contaminant transfer from the sediment is clear; these animals feed in the surface sediment. A link to the sediments is less clear for animals that do not feed on soft-bottom infauna, for example, kelp bass (Paralabrax clathratus), or for species that are transients in the area or live some distance from the contaminated sediment. This is particularly true when attempting to explain the elevated levels of DDTs and PCBs in birds and sea lions from the Channel Islands.

In addition to the sediments near the Whites Point Outfall, other sources of contamination to the coastal biota of California have been suggested: migrant birds (e.g. Hunt et al. 1986), migrant fish, runoff from agricultural fields in which Kelthane was used (San Joaquin Valley, Hunt et al. 1986), residual DDT in Salinas Valley soils (Risebrough and Jarman 1985), illegal DDT use, global fallout (Hunt et al. 1986) and other coastal sources of PCBs. Although the

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[1] In Figures 1-1 and 1-2, the data are presented as averages for each of several spatial segments, or regions. The spatial segments are described in Section 2 of this report.
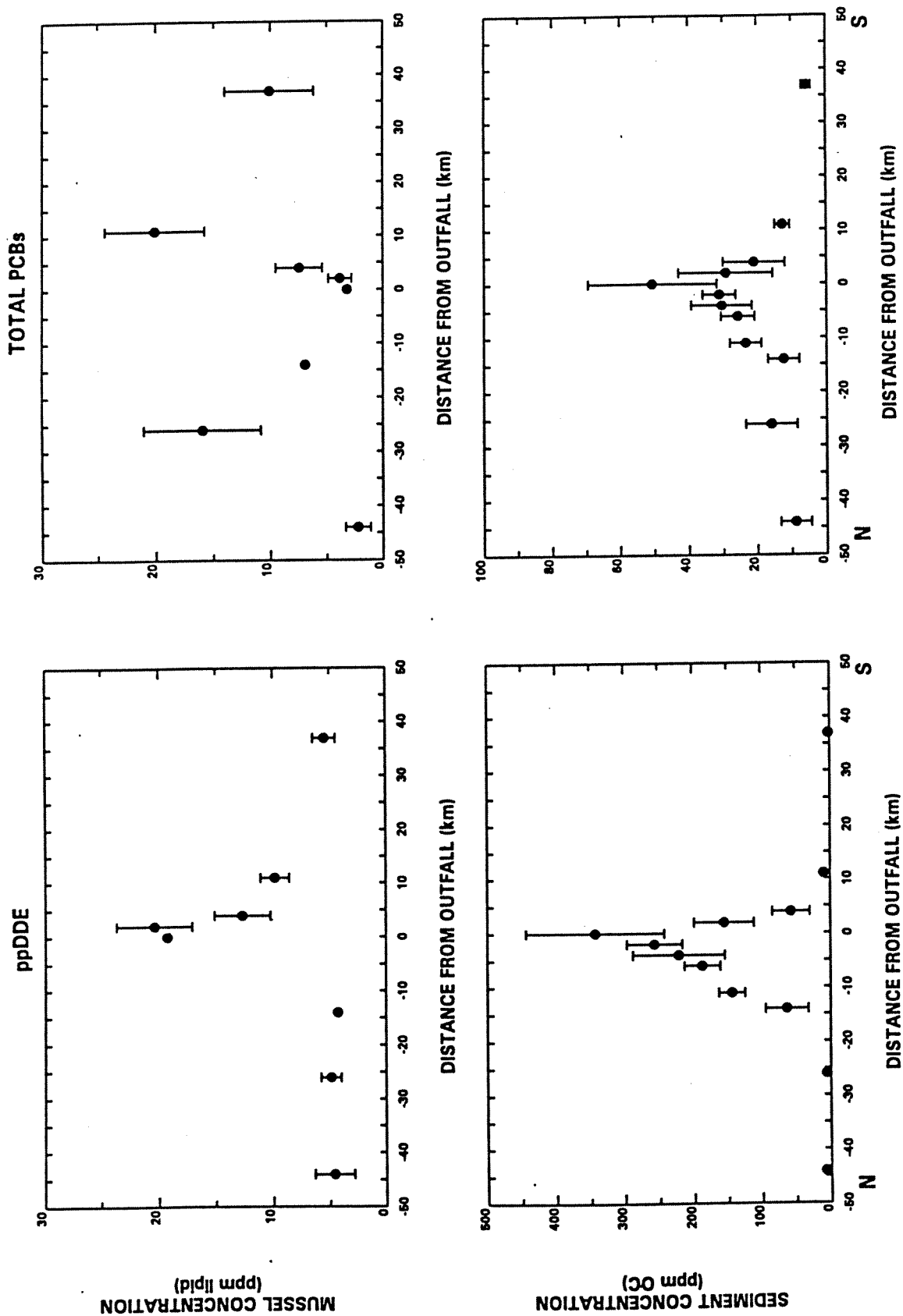
Figure 1-1. Alongshore (north-south) spatial profiles of p,p'DDE and PCB concentrations in mussels (ppm lipid) and surface sediments (ppm organic carbon) from the Southern California Bight. Data are arithmetic means and +/- 2 standard errors of the mean. Concentrations are plotted as a function of distance from the Los Angeles County outfall (segment 8; kilometer 0).
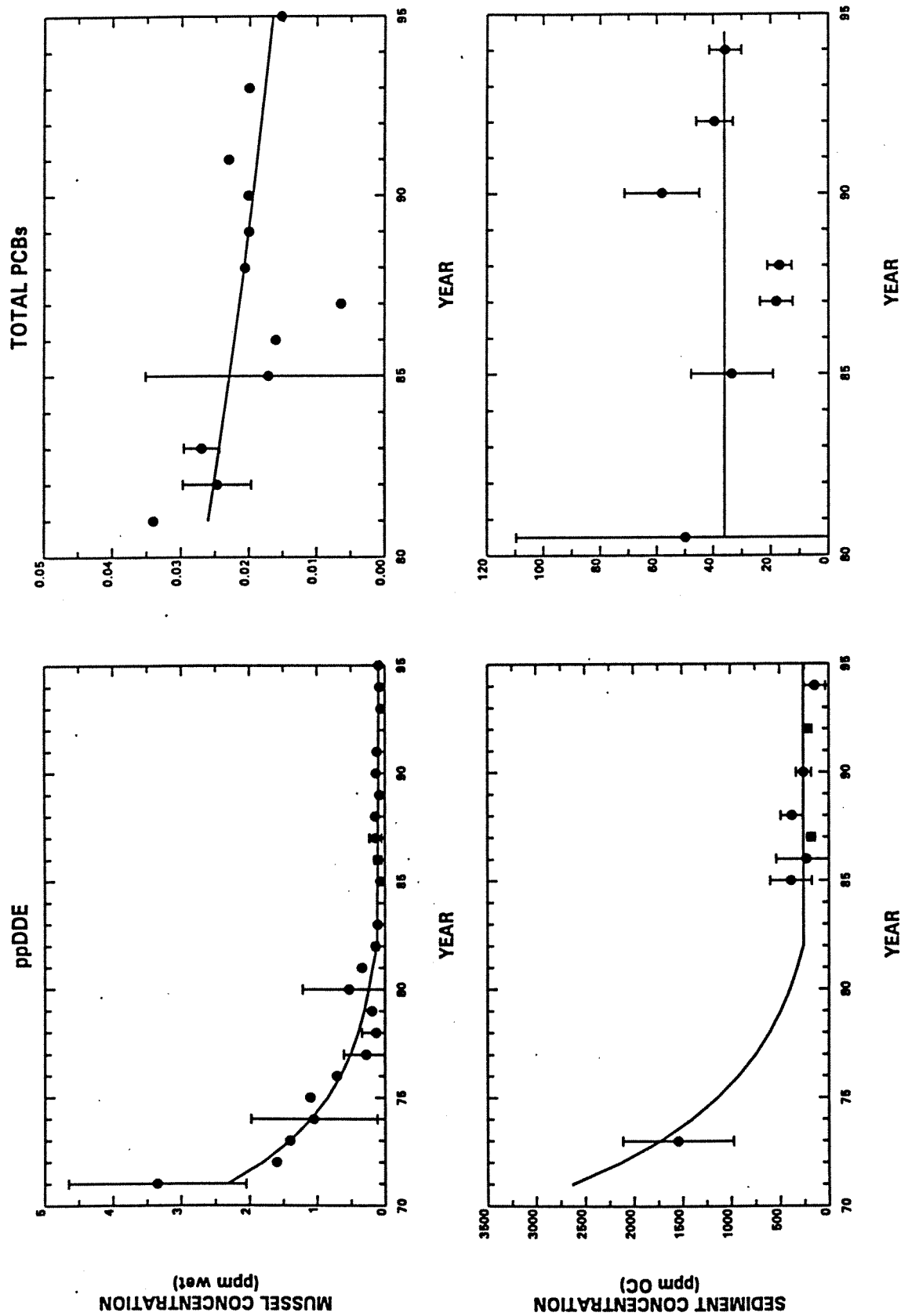
Figure 1-2. Temporal profiles of annual average p,p'DDE and PCB concentrations in mussels (ppm wet) and surface sediment (ppm organic carbon) within 2 km of the Whites Point outfall. Data are arithmetic means +/- 2 standard errors of the mean of measurements from samples collected within HydroQual segment 7 (-2 km, sediment) and segments 8 and 9 (0-2 km, mussels). Lines indicate the time trends used by the models.

spatial patterns of DDTs along the West Coast from Mexico to Northern California exhibit a distinct peak at Palos Verdes in sediments and in mussels (see Figures 1-3 and 1-4), other smaller peaks do exist for mussels: for example, several peaks in PCB concentrations are seen in the Southern California Bight. However, the absence of elevated concentrations in sediments close to the locations of these peaks is an indication that the mussel peaks probably reflect localized contamination. In addition, loading of DDT from agricultural runoff was dwarfed by the loading from the Whites Point outfall in the 1970s (Stout and Beezhold 1981).

The purpose of this study is to provide further quantitative evidence as to whether the contamination currently observed in the animals not directly exposed to the sediment plume may have originated from sources other than the Palos Verdes Shelf sediments. The validity of this argument is tested by using quantitative models of trophic transfer to estimate the extent of contamination likely to result from exposure within and outside of the Palos Verdes area.

1.2 GENERAL APPROACH

The general approach used in this study consisted of estimating the route and magnitude of p,p'DDE and PCB transfer to a set of species of concern. These species include: three fish, white croaker (Genyonemus lineatus), Dover sole (Microstomus pacificus) and kelp bass (Paralabrax clathratus); three birds, bald eagle (Haliaeetus leucocephalus), peregrine falcon (Falco peregrinus) and double-crested cormorant (Phalacrocorax auritus); and one marine mammal, the California sea lion (Zalophus californianus). We attempted to answer specific questions regarding the p,p'DDE and PCB contamination in these species.

For the white croaker, Dover sole, and kelp bass. Is food web transfer from the local, highly contaminated sediments and/or water column both necessary and sufficient to account for the p,p'DDE and PCB concentrations observed in the local populations of these fish? That is, does transfer from local sediments and waters account quantitatively for the observed fish levels? Could their contaminant levels have been achieved by exposure to levels characteristic of areas beyond the Palos Verdes region?

We began by examining the existing contaminant data in sediment, mussels and fish. This data analysis included evaluation of spatial and temporal trends. Local (Palos Verdes), regional (southern California) and continental (Mexico to Alaska) scales were used in this analysis. The goals of this examination were: 1) to establish the p,p'DDE and PCB
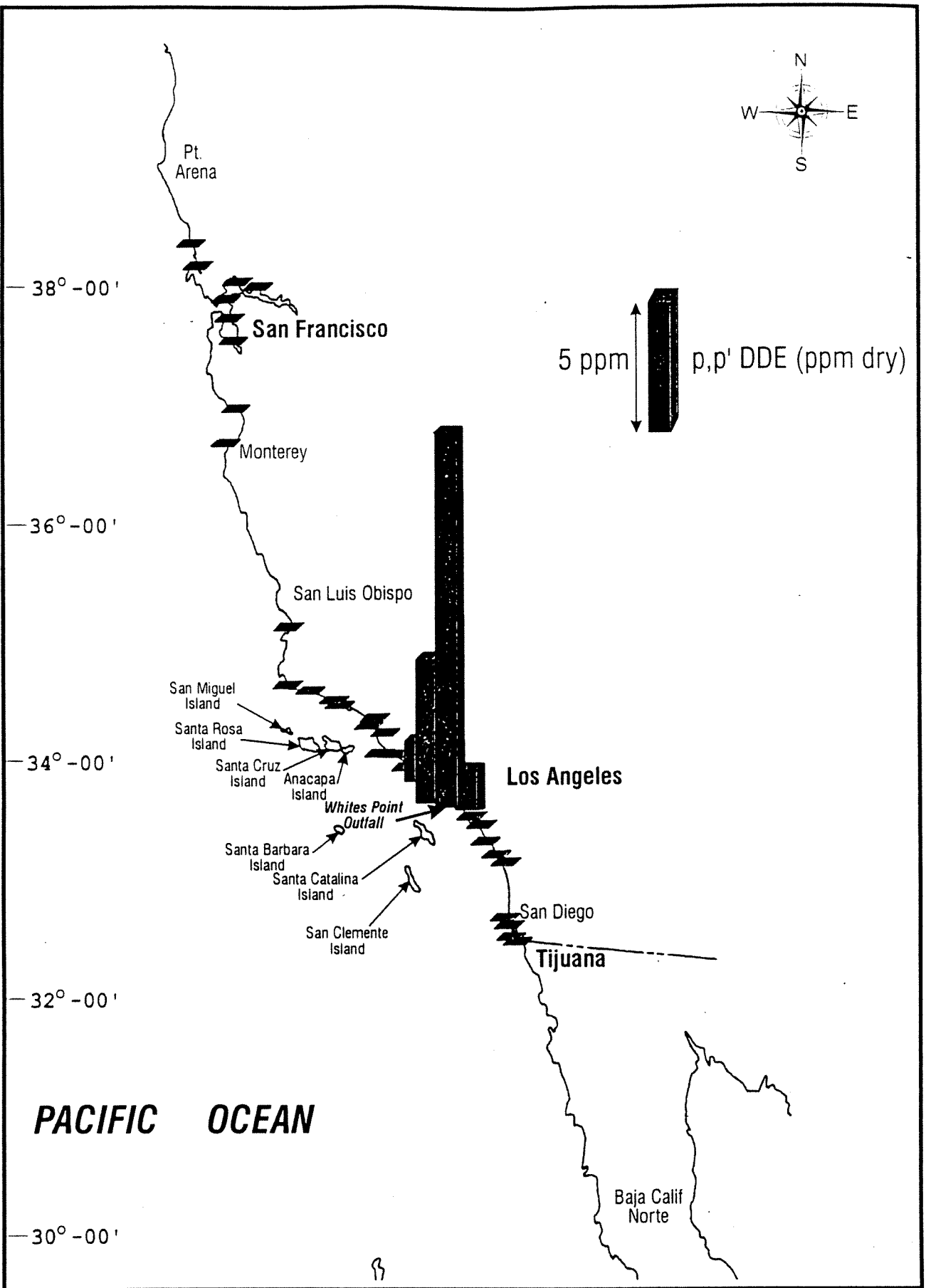
Figure 1-3
Spatial profile of average p, p' DDE
concentrations (ug/g Dry) in Surficial Sediments
collected between 1985 and 1995
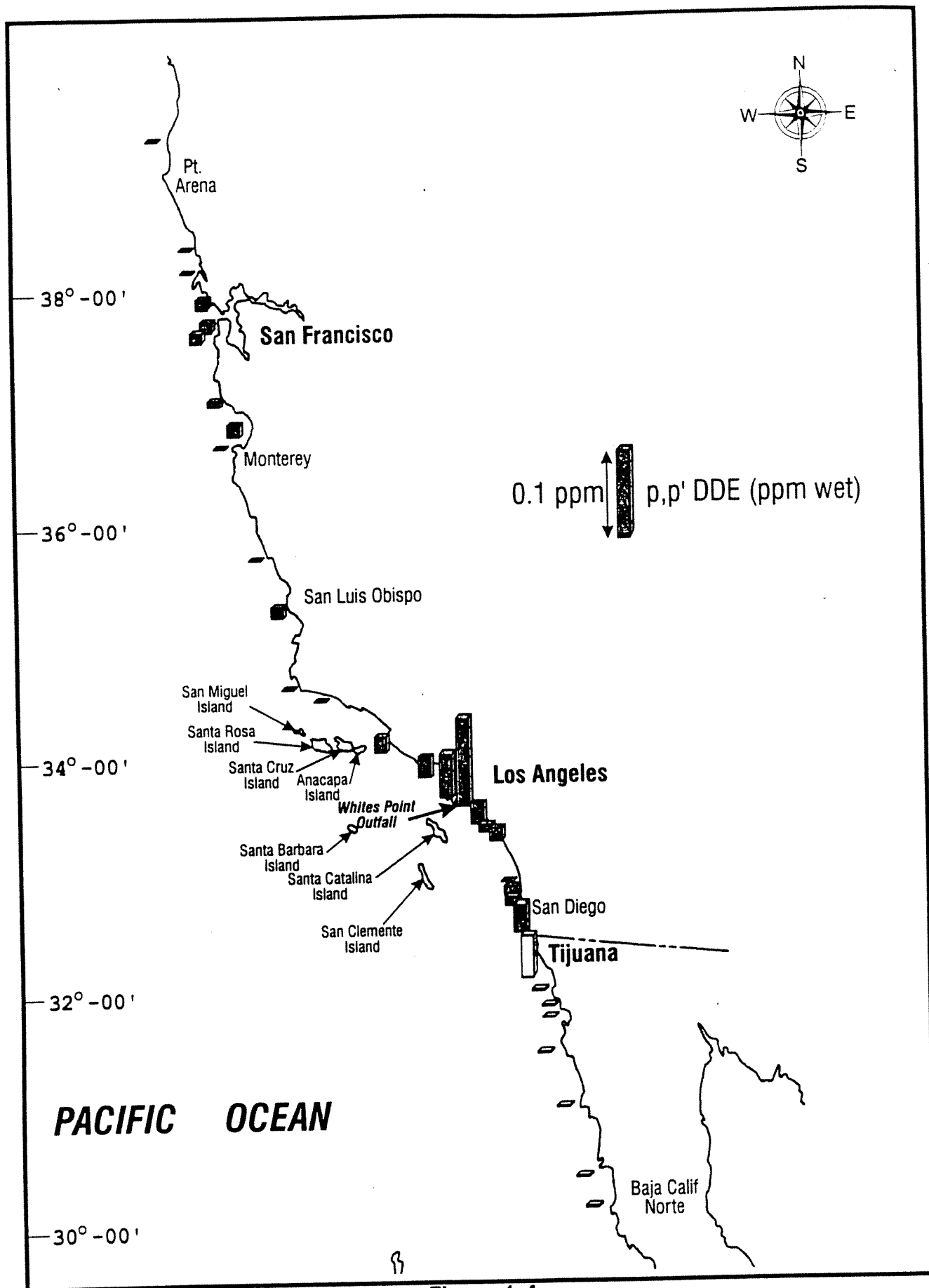
Figure 1-4
Spatial profile of average p,p' DDE concentrations (ppm wet)
in mussels collected in California between 1985 and 1995. Mussels
in Mexico were collected in 1983. (Martin et al., 1984)

concentrations to which the biota may be exposed, 2) to look for evidence of sources, and 3) to determine the degree of correlation between water, sediment and biota p,p'DDE and PCB concentrations.

We proceeded by developing models to quantify the relationships between contaminant levels in exposure sources and contaminant levels in each species of interest. All of the models were developed using the same basic theory, and model parameter values were determined in a consistent fashion. These models were then used in similar ways for each species: 1) relevant biological knowledge was used to define the food web pathway from sediment and water to the fish; 2) the rate and extent of p,p'DDE and PCB transfer through the pathway was quantitatively estimated; and 3) based on the model calculations, the p,p'DDE and PCB concentrations in the fish that were quantitatively attributable to the elevated sediment and water concentrations in the Palos Verdes area were compared to observed concentrations.

For sea lions. To account for the p,p'DDE and PCB concentrations observed in populations of the Southern California Bight, what must have been the concentrations in their prey? Do these estimated dietary concentrations indicate that they must have received much of their contaminant loads from the Palos Verdes area?

The approach was similar to the approach used for the fish. 1) Biological and toxicokinetic knowledge was used to develop relationships between ingestion of prey contaminated with p,p'DDE and PCB and accumulation of these chemicals. 2) Prey p,p'DDE and PCB concentrations were estimated from these relationships and from observations of body burdens in the sea lions. 3) These prey concentrations were compared with concentrations observed in known prey species on the shelf to evaluate if an association with the area of high concentration (i.e., Palos Verdes) would be required to accumulate the observed concentrations.

For the peregrine falcon, bald eagle and double-crested cormorant. Is food web transfer from fish, birds and marine mammals of the Southern California Bight necessary and sufficient to account for p,p'DDE and total PCB concentrations observed in the local populations of these birds?

The approach was as follows. 1) Relevant biological knowledge and direct field observation were used to determine the composition of the diet and the extent of movement of each species of interest. 2) Concentrations of p,p'DDE and total PCBs in the prey were

estimated from field data. 3) The rate and extent of contaminant transfer from prey to egg was estimated using a bioaccumulation model. 4) Observed and computed contaminant levels in eggs of the species of interest were compared in order to evaluate whether the measured contaminant levels are quantitatively consistent with our understanding of the birds' feeding behavior and movement in the Southern California Bight. In addition, the proportion of the contaminant dose that originates within the Bight was estimated for the peregrine falcon and the bald eagle, and the impact of alternative assumptions concerning double-crested cormorant feeding behaviors on egg contaminant levels were explored.

1.3 BIOACCUMULATION MODEL

For the fish. A model of the bioaccumulation process was used to quantify the pathway from sediment and water to fish. The bioaccumulation model quantifies the rates of uptake and loss of contaminants by animals in a structured food web. For example, the model for the white croaker or Dover sole includes calculation of the accumulation of p,p'DDE and PCB in benthic invertebrates as the invertebrates ingest contaminated sediment and respire contaminated water. The white croaker and Dover sole in turn accumulate the chemicals from the invertebrates by ingesting them. Although our interest lies with the fish, the lower levels of the food web are necessary components of the model because of the successive accumulation that occurs as p,p'DDE and PCBs move through the food web. The food webs of white croaker, Dover sole and kelp bass are shown in Figure 1-5.

The model includes all of the major processes by which animals take in, store and eliminate chemicals. These include transfer of the ingested chemicals across the gut wall, transfer across the gill surface of the chemicals in ventilated water and in blood passing through the gill, storage of the chemicals in the animal's fat tissue, and elimination of the chemicals by excretion (from kidney or liver). The full life cycle of the animal is considered. The transfer of contaminant between the water or sediment and the lower levels of the food web (i.e., plankton and macroinvertebrates) is modeled as a simple partitioning process.

The structure of the model is formed by the food web along with the mass and energy balance equations that describe the predator-prey relationships and the transfer of contaminant. Use of this structure to estimate contaminant concentrations in the fish involves specifying values for the various parameters in the equations and the concentrations of contaminants in the water column and sediment. The parameter values were derived from peer reviewed experiments
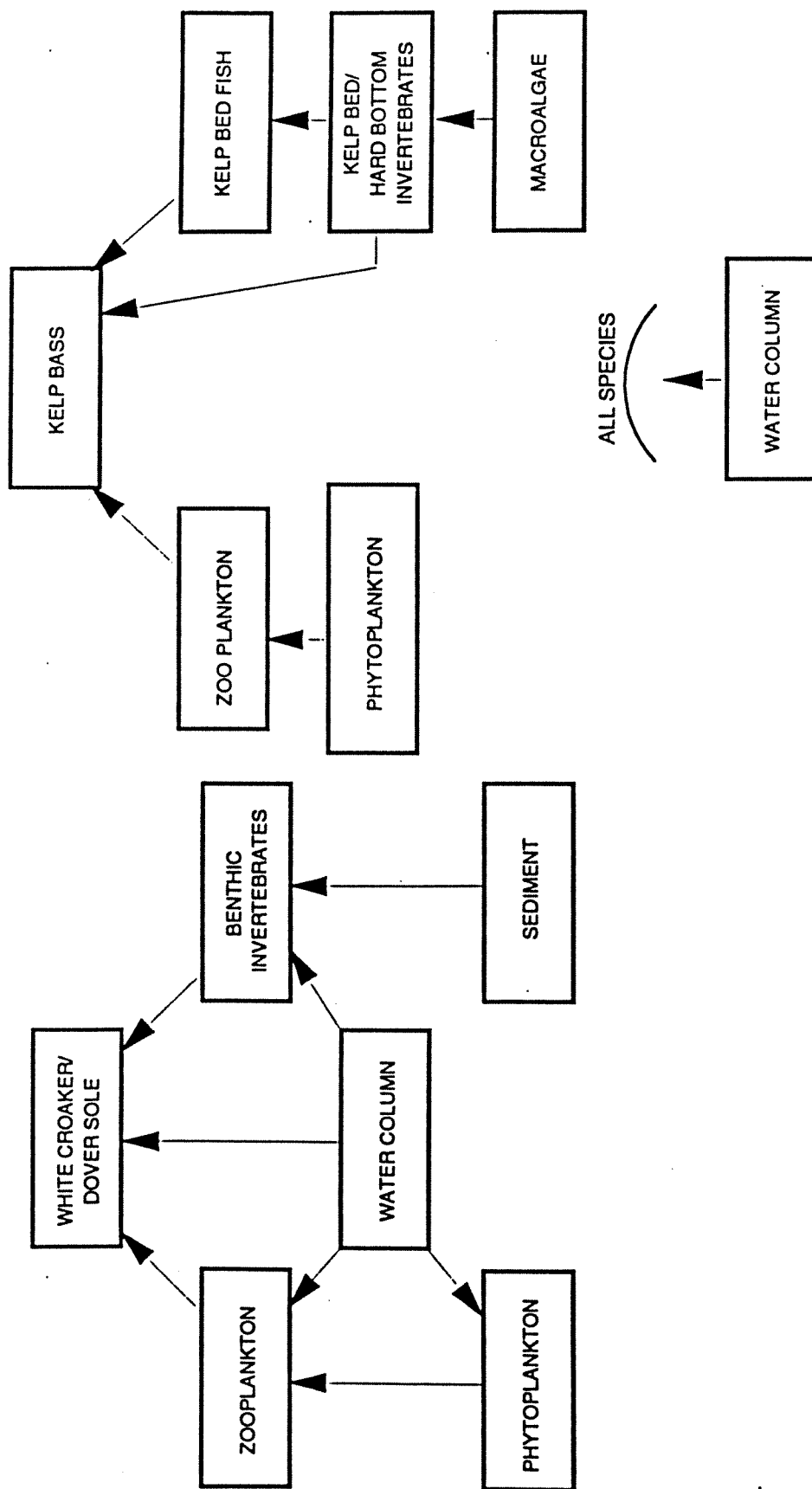
Figure 1-5. Food webs of the white croaker, Dover sole and kelp bass.

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published in the open literature and/or site specific data. They are the best estimates currently available. The water column p,p'DDE and PCB concentrations were derived from concentrations measured in mussels and relationships between mussel and water concentrations estimated from field studies. Measured surface sediment p,p'DDE and PCB concentrations were used directly. Values for exposure levels and toxicokinetic parameters were identical for all three fish models; thus, the models were not calibrated independently. Using best estimates and not adjusting them to "fit" the data increases the strengths of the conclusions.

We have successfully used this model framework to describe the relationship between contaminant concentrations in aquatic food webs and sediment and water in many systems. A list of these applications is shown in Table 1-1.

Table 1-1. Previous Applications of the Bioaccumulation Model

System	Contaminant	Food Web Leading to	Reference
Lake Michigan	PCBs	lake trout	Thomann & Connolly 1984
Lake Ontario	PCBs	lake trout	Connolly & Thomann 1992
James River Estuary	Kepone	striped bass	Connolly & Tonelli 1985
Hudson River Estuary	PCBs	striped bass	Thomann et al. 1991
New Bedford Harbor	PCBs, Cd, Cu, Pb	winter flounder, lobster	Connolly 1991
Green Bay	PCBs	walleye, brown trout	Connolly et al. 1992
Hudson River	PCBs	largemouth bass	Glaser and Connolly, 1997 unpublished

For sea lions. The model of contaminant accumulation in female sea lions was developed in similar fashion to that of the fish; equations derived from the principles of energy and mass conservation were parameterized using best available information obtained from the open literature, including general and site specific data, as well as the results of the expert reports produced as part of the Southern California Bight Damage Assessment. The sea lion model differed from the fish model in that uptake and loss across the gill were not included; transfer from mother to pup via lactation was found to be the dominant loss mechanism for female sea lions.

For birds. The bird models differed from the sea lion model in that lactation was not included, and transfer from mother to egg was included. Also in contrast to the sea lions, metabolic modification of p,p'DDE and PCBs and subsequent excretion of the metabolic products was found to be the most important loss mechanism. Values for the excretion rates and

the other model parameters were determined in a consistent fashion for all three bird species using published data. Thus, as for the fish models, the bird models were not calibrated independently. Data concerning dietary composition, contaminant levels in the prey and contaminant levels in eggs collected as part of the Southern California Bight Damage Assessment were used.

1.4   RESULTS

1.4.1   Contamination of Sediments, Water and Fish of the Southern California Bight

Analyses of measured contaminant levels in sediments, water and fish of the Southern California Bight indicate that the Palos Verdes shelf represents the most important sediment source of p,p'DDE and PCBs in the Southern California Bight. Measurements of p,p'DDE in sediments over the period from 1970 to 1995 indicate that p,p'DDE contamination in sediments on the Palos Verdes shelf has been greater than one order of magnitude higher than at any other sampled location along the California Coast. Measurements of p,p'DDE in mussels indicate that from the early 1970's to the early 1980's levels in mussels on the Palos Verdes shelf were also more than one order of magnitude greater than neighboring areas, based on data collected from Mexico to Alaska. Between the early 1970's and the late 1980's, p,p'DDE concentrations in mussels on the shelf declined more rapidly than concentrations in neighboring areas. By the late 1980's, the relative size of the peak was smaller, and mussel concentrations in some areas of California north of Point Conception approached those on the shelf.

The declines in sediment and mussel p,p'DDE and PCB concentrations are asymmetrical around the peak at Palos Verdes, dropping off rapidly to the south and more gradually to the north, reflecting the effect of dominant bottom currents in the region. Sediment concentrations drop more quickly than do mussel concentrations, reflecting the greater dispersion of contaminants in the water column.

White croaker p,p'DDE concentrations exhibit a spatial pattern similar to concentrations observed in the sediment. Dover sole and kelp bass follow the trend in sediment concentrations to a lesser extent. The data for these species are qualitatively similar to that of the mussels, suggesting that, in contrast to the croaker, they may have derived a portion of their contamination from food sources in the water column.

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Concentrations in sediments and mussels declined steadily from 1970 to the early to mid-1980's. Concentrations in fish showed great variability during the 1970's, and it is not always possible to see the declines that were seen in the sediments and mussels. White croaker, Dover sole, and kelp bass, sediments and mussels p,p'DDE concentrations have not exhibited a trend since the mid-1980's.

1.4.2 Contamination in Palos Verdes Fish

The quantitative bioaccumulation modeling in fish was focused on the Palos Verdes Shelf. First, the models were used to estimate concentrations in the fish that would result from exposure to water column and sediment concentrations in the vicinity of the outfall. The lines shown in Figure 1-2 indicate the sediment exposure concentrations and the concentrations in mussels that were used to compute water column exposure concentrations. Fish p,p'DDE and PCB concentrations computed by the model were then compared to concentrations observed in fish collected in the vicinity of the outfall (Figure 1-6). In all cases the models indicate that the fish p,p'DDE and PCB concentrations expected from exposure to p,p'DDE and PCBs in the local environment are consistent with observed concentrations.[2]

Thus, based on these model results as well as spatial patterns observed in the data, white croaker, Dover sole and kelp bass populations in the vicinity of the outfall accumulate contaminants from local sediments and water; that is, they do not move over great distances. Further model simulations suggest that the p,p'DDE concentrations in all three species are consistent with local sediment and water column exposure levels throughout the region extending from 14 km north of the outfall to 11 km south of the outfall, a region over which exposure levels vary several-fold. In addition, even if kelp bass spend one-fourth of their time in an area of lower contamination (the area of Santa Catalina Island), their calculated levels are still consistent with levels observed in fish caught on the Palos Verdes Shelf. However, if kelp bass spend all of their time in the area near Santa Catalina Island, then their calculated levels are much lower than the Palos Verdes data. In other words, the p,p'DDE and PCB concentrations in the sediments and water of the Palos Verdes Shelf are necessary and sufficient to account for all of the p,p'DDE and PCBs observed in fish living on the shelf. That is, the water and sediment p,p'DDE and PCB concentrations observed outside the Palos Verdes Shelf are too low

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[2]Model/data comparisons are presented on a lipid basis in Figure 1-6. Wet weight-based comparisons are presented in Section 3.

ppDDE

TOTAL PCBs

FILLET CONCENTRATION (ppm lipid)

WHITE CROAKER

WHITE CROAKER

DOVER SOLE

DOVER SOLE

KELP BASS

KELP BASS

YEAR

YEAR
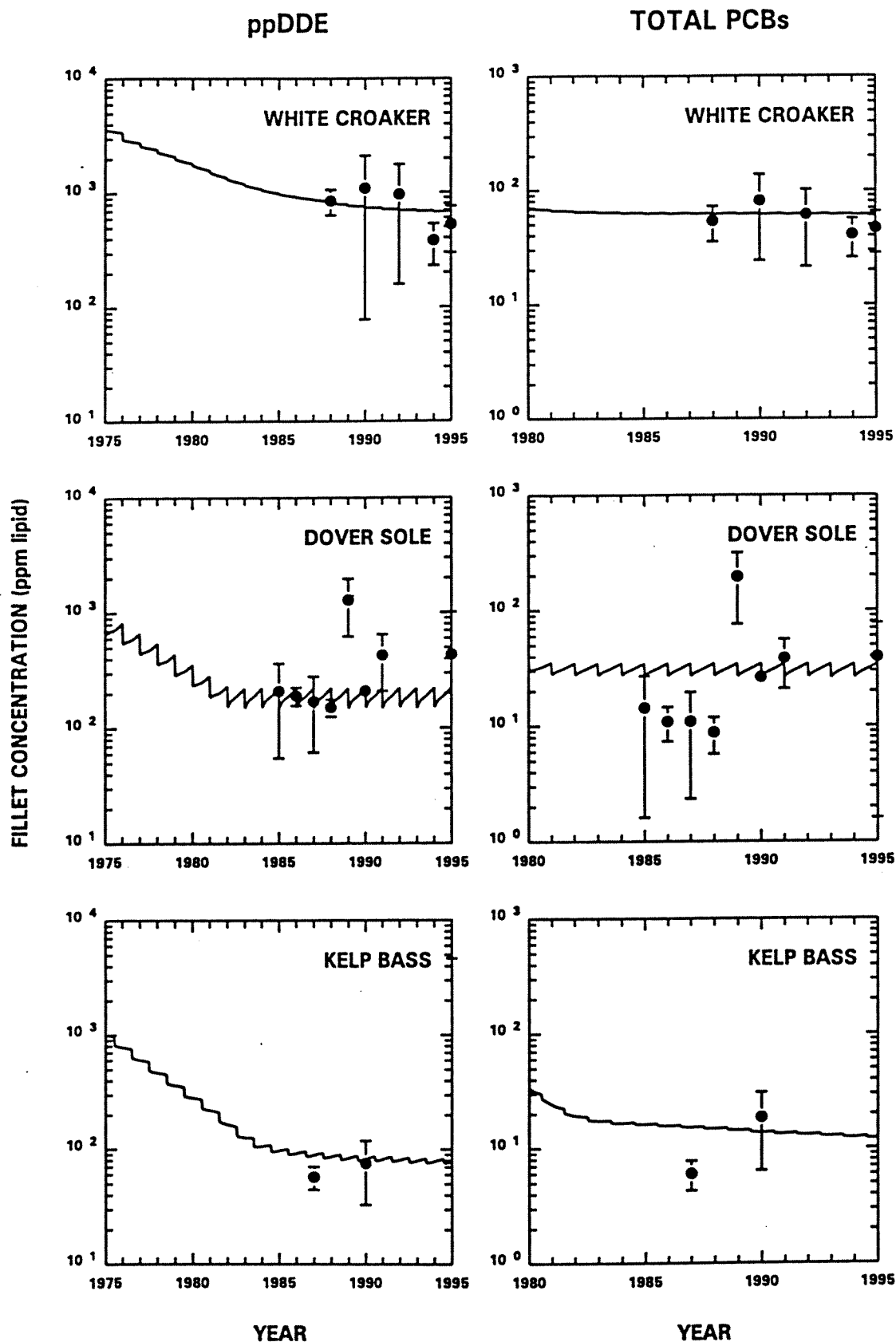
Figure 1-6.   Comparison of predicted and observed p,p'DDE and PCB concentrations (ppm lipid) in white croaker, Dover sole and kelp bass from the vicinity of the Whites Point outfall. Data are arithmetic means and +/- 2 standard errors of the mean.

to be responsible for the contamination of these fish.

We examined the uncertainty of the models to determine if inexact knowledge of model parameters would affect these conclusions. The uncertainty in parameter values resulted in a range of computed fish concentrations that overlapped and were similar to the confidence intervals of the data, suggesting that the conclusions are not invalidated by model uncertainty.

Dietary p,p'DDE and PCB Concentrations of San Miguel Island Female Sea Lions

Dietary concentrations were chosen to produce an empirical best fit to the sea lion p,p'DDE and PCB concentrations (Figure 1-7). In these simulations, the temporal change in sea lion dietary p,p'DDE and PCB concentrations was assumed to be similar to that observed in the mussels and sediment (see Figure 1-2), i.e., an exponential decline from 1970 to the mid-1980s and constant thereafter. These dietary concentrations were then compared to concentrations observed in prey species aggregated into three broad regions: the Palos Verdes Shelf, the area of the Southern California Bight north of the shelf and Santa Catalina Island. Most of the data included in the area north of the shelf are for fish collected in nearshore areas on the Santa Monica Shelf within 50 km of the Los Angeles County outfall. By aggregating data this way we have attempted to isolate three levels of contamination. Both the Palos Verdes region and the region north of Palos Verdes have elevated fish contaminant levels in a pattern consistent with a source at the Whites Point Outfall. Prey species concentrations from Santa Catalina Island are much lower than either of these areas and possibly representative of concentrations in the vicinity of San Miguel Island.

Note that the results of the model best represent the dietary concentrations of adult females rearing pups. The model is less certain for the juvenile animals for three reasons: 1) the major route of contaminant loss differs between juveniles and parenting females (i.e., excretion versus lactation) and excretion loss is less well known than lactation loss; 2) the data for the younger animals (premature parturient females) are more variable than that of the older animals (full-term parturient females), precluding discrimination between various exposure scenarios; and, 3) the contaminant concentrations in the parenting females do not depend on exposure prior to their first lactation cycle, because nearly all of the accumulated body burden
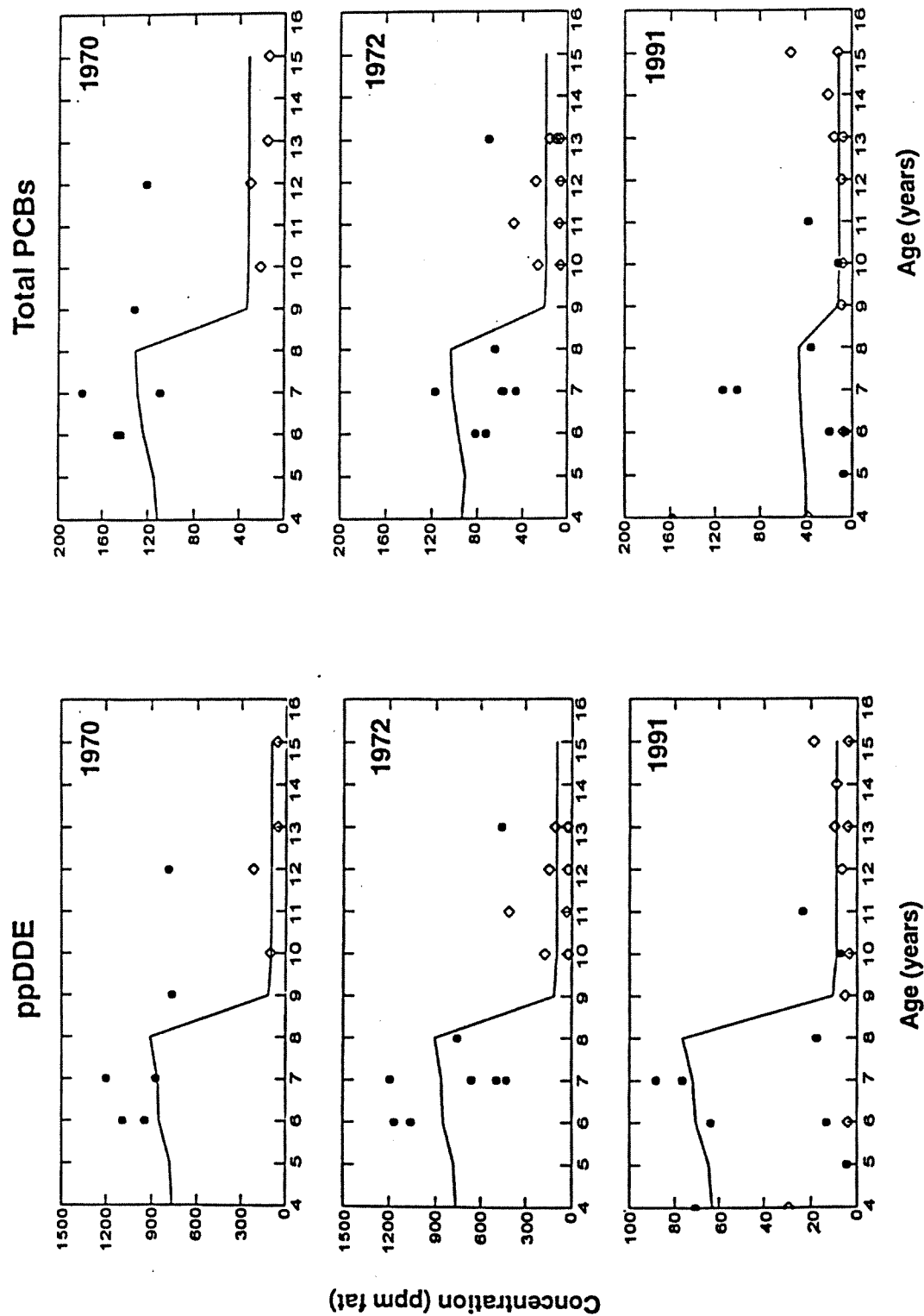
Figure 1-7. Comparison of predicted and observed p,p'DDE and PCB concentrations (ppm fat) in female sea lions from San Miguel Island. Solid circles are premature parturient animals. Open diamonds are full-term parturient animals.

is lost during nursing (i.e., the model can reproduce the data for the full-term parturient females independent of the exposure scenario assumed for the juvenile animals).

Comparisons of computed and observed prey p,p'DDE and PCB concentrations are presented in Figure 1-8. The model indicates that the concentrations observed at a low-level contaminant site (Santa Catalina Island) are insufficient to account for the p,p'DDE and PCB concentrations measured in the female sea lions. By contrast, the prey contaminant levels in the regions closer to the outfall are sufficient to account for the measured concentrations. Although we cannot estimate the pattern of feeding resulting in the required prey contaminant level (e.g., feeding across the concentration gradient between the outfall and San Miguel Island versus feeding consistently in the Santa Monica Shelf area), the average sea lion prey had elevated p,p'DDE concentrations. A similar pattern is seen for PCBs although the association with elevated concentrations is less clear. The source of these contaminants was most probably the Whites Point outfall. This is so because: 1) adult females rearing pups spend most of the year foraging from San Miguel Island (a consequence of a 9 to 10 month nursing commitment) and, 2) the only area with observed contaminant concentrations that are clearly sufficient to yield the levels observed in the sea lions and that is within foraging range of the island is the Palos Verdes Shelf.

Dietary p,p'DDE and PCB Concentrations in Peregrine Falcons, Bald Eagles and Double-Crested Cormorants

The strategy for the birds was to use field-measured dietary composition and prey contaminant levels along with the bioaccumulation model to compute egg levels in the species of interest. Then, these computed levels were compared with field-measured egg levels in the predators. The relationship between the computed and measured egg levels provides an assessment of the accuracy with which the dietary composition characterizes the contaminant sources to the bird species.

Computed p,p'DDE and total PCB levels in the eggs of the peregrine falcon and bald eagle are plotted against measured levels in Figure 1-9. The horizontal error bars in Figure 1-9 represent +/- two standard errors of the mean of the data. The vertical error bars present the range of model results based upon a key uncertainty in the model, the fraction of dietary lipid in the eggs. The purpose of the error bars is to provide an indication of the uncertainty associated with the measured and computed egg levels in order to compare the results with the
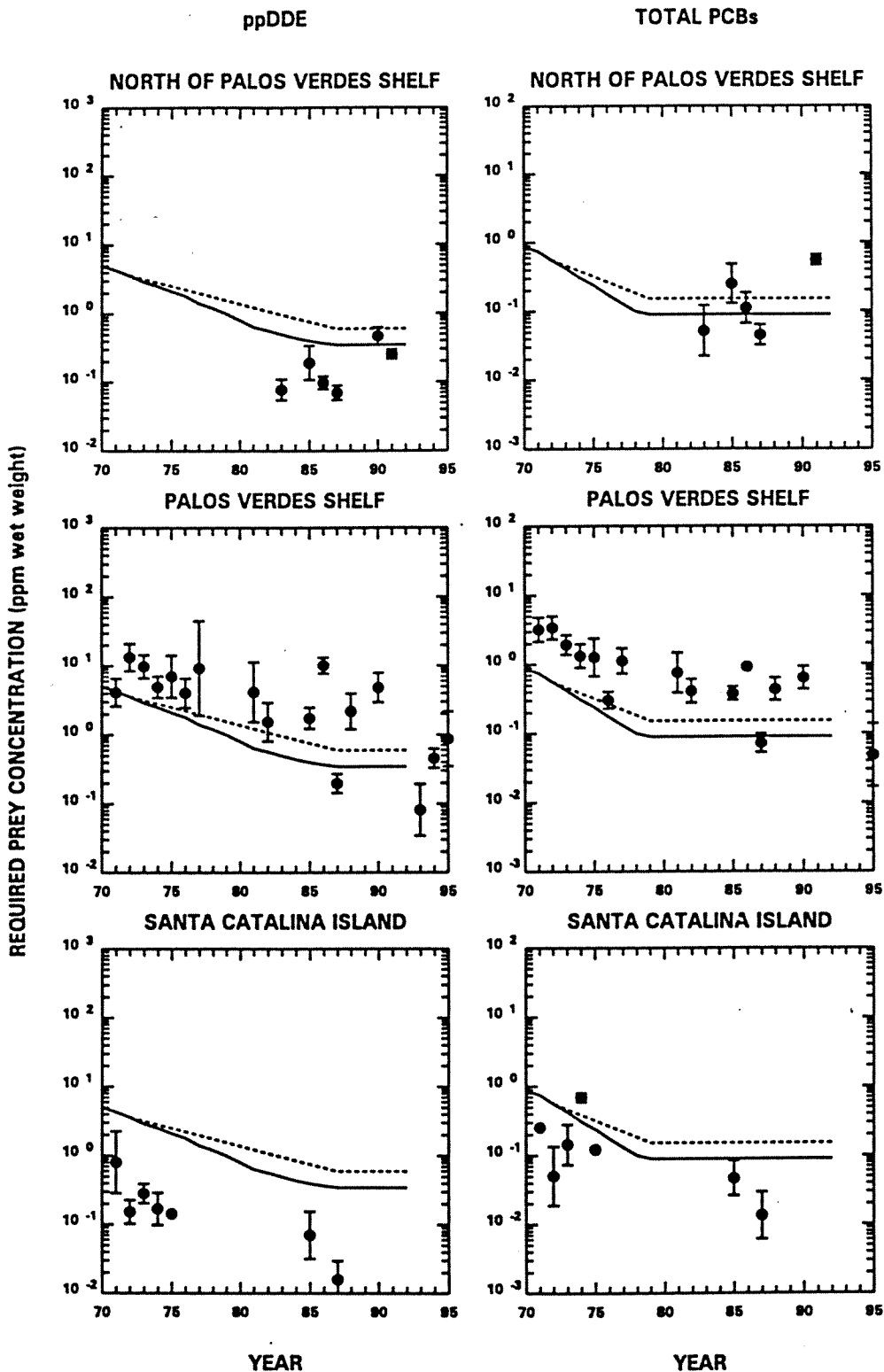
Figure 1-8.   Comparison of dietary p,p'DDE and PCB concentrations (ppm wet weight)
necessary to achieve the contaminant levels observed in San Miguel Island
female sea lions. Data are arithmetic means and +/- 2 standard errors of the
mean. The two lines reflect old GERG and new GERG measurements of
contaminant levels in the sea lions (see Section 4).
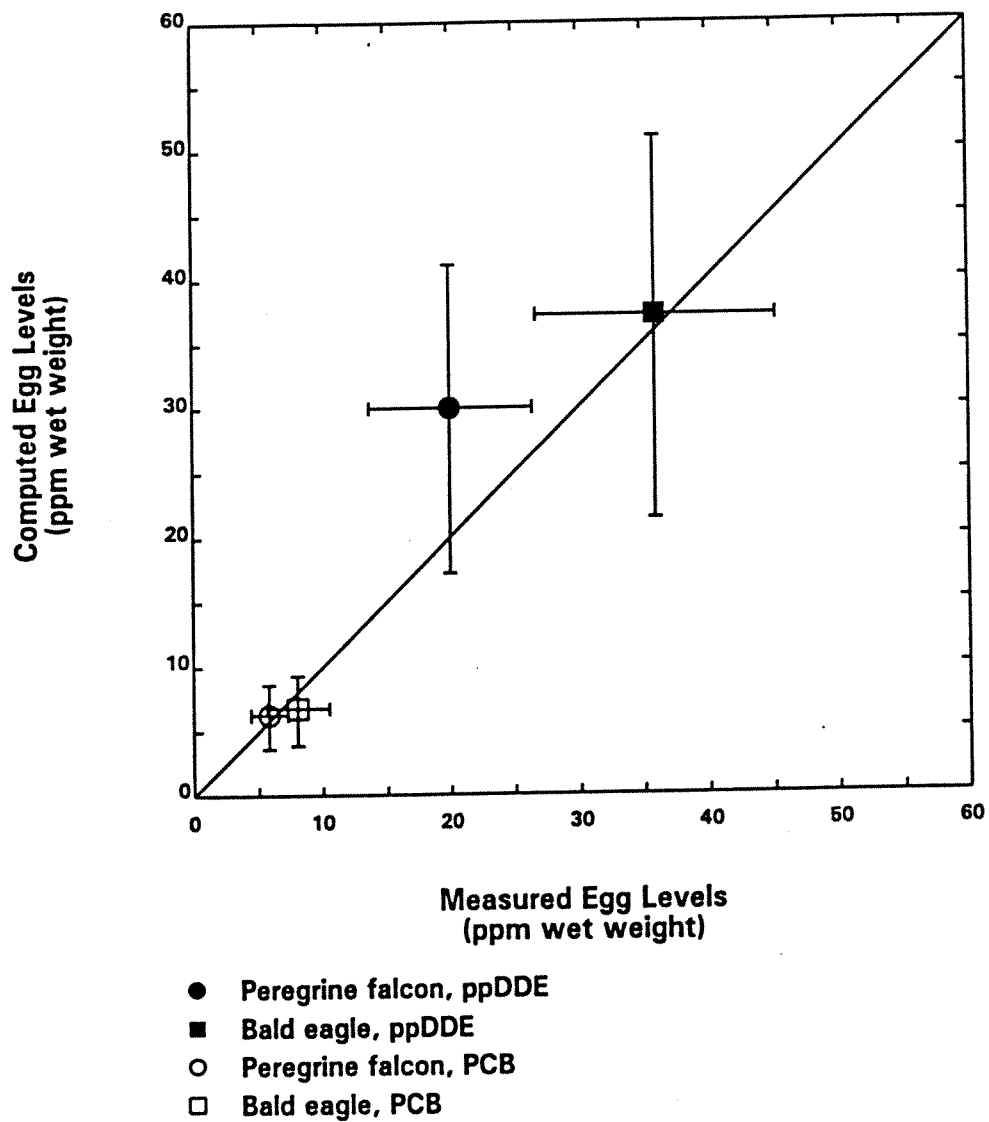
Figure 1-9. Computed and observed contaminant concentrations in the eggs of the peregrine falcon and bald eagle from the Southern California Bight (ppm wet weight). Horizontal error bars represent +/- two standard errors of the mean. Vertical error bars represent the range of the fraction of dietary lipid in the eggs.

one-to-one line. In 3 cases, the computed and measured egg levels are within 20 percent of each other, and in one case, within 50 percent. Thus, the dietary composition developed from field measurements during the Southern California Bight Damage Assessment characterizes the contaminant sources to these two bird species realistically.

As for the bald eagle and peregrine falcon, the computed and measured egg contaminant levels are similar for the double-crested cormorants breeding on Anacapa Island (Figure 1-10, top panel). In contrast, computed egg levels in the cormorants from Santa Barbara Island are considerably greater than measured levels, indicating that these birds probably spend less time feeding in the more contaminated regions of the Southern California Bight than hypothesized. To explore this further, an additional set of simulations were performed assuming the birds fed exclusively in the vicinity of their breeding island all year (Figure 1-10, bottom panel). Now, the computed egg levels for Anacapa Island are less than measured levels, indicating that some degree of exposure to the more highly contaminated regions of the Southern California Bight is necessary to achieve the levels observed in these cormorants. The computed levels in the Santa Barbara cormorants are still greater than the measured levels. Therefore, these birds are likely to be feeding in even less contaminated areas than the region within 50 km of Santa Barbara Island.

The amount of each contaminant that originates outside of the Southern California Bight and finds its way to the bald eagle and peregrine falcon *via* migratory prey was also quantified. To explore this, the dose received by the bald eagle and peregrine falcon was apportioned between sources within and outside of the Bight. The potential importance of sources external to the Bight was probably overestimated by assuming that all migratory bird prey spend 50 percent of the year outside of the Southern California Bight and that the contaminant concentrations to which these prey are exposed outside of the Bight are the same as within the Bight.

Considerably more than 50 percent of the contaminant dose to the peregrine falcon and bald eagle originates within the Southern California Bight. Based on the measured prey contaminant levels and dietary compositions, at least 75 percent of the p,p'DDE and 69 percent of the PCBs in peregrine falcon eggs from the Channel Islands are accounted for by local prey, and at least 92 percent of the p,p'DDE and PCBs in bald eagle eggs from Santa Catalina Island are accounted for by local prey. These percentages represent the minimum contribution of sources within the Southern California Bight to the bald eagle and peregrine falcon.
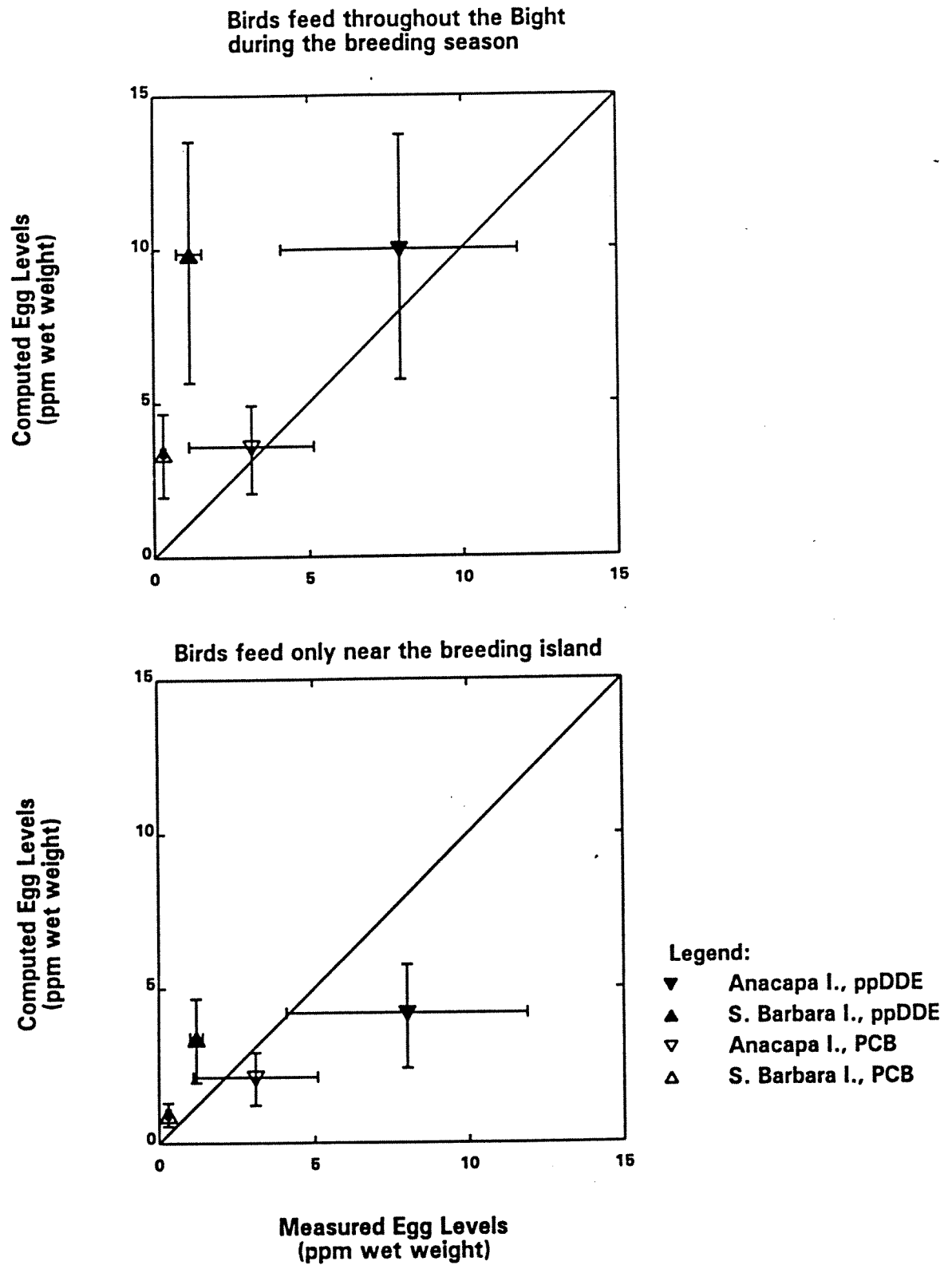
Figure 1-10. Computed and observed contaminant concentrations in the eggs of the double-crested cormorant from the Southern California Bight (ppm wet weight). Horizontal error bars represent +/- two standard errors of the mean. Vertical error bars represent the range of the fraction of dietary lipid in the eggs.

1.5   FURTHER APPLICATIONS OF THE MODELS

The model framework has been applied to three species of fish, sea lions, and three species of birds as part of this effort. The framework is general, it could also be applied to other species in the Southern California Bight.

Once a model is developed for a given species, it can be used to answer a variety of questions. For example, the model allows for the prediction of contaminant levels in the species of interest given various temporal or spatial patterns of exposure levels. Thus, it is possible to project future contaminant levels in a fish species based on projected future sediment and water contaminant levels. In addition, it is possible to perform the reverse calculation, that is, to estimate the contaminant level in the sediments and the water column that would produce a specified concentration in fish tissue.

Projection of future concentrations in sediments and water column. To illustrate this, the white croaker, Dover sole and kelp bass models were used to estimate the concentrations in the sediment and water column that would produce specified levels in fish tissue. In this case, the specific values selected were those determined by Pollock *et al.* (1991) for levels of total DDTs and total PCBs in fish tissue based on potential human cancer risks (0.1 ppm wet weight in the fillet for both contaminants). The models for the three fish species were used to establish relationships between levels of contaminants in the fish and levels in the sediments and water column. These relationships were then used to calculate the levels of contaminants in sediments and water that would produce these specified levels of contaminants in the fish. For the white croaker and Dover sole models, sediment levels are reported, because the food web, which is tied to the sediment, is the dominant source of contaminants to these fish. For the kelp bass model, the water column levels are reported, because the food web is tied to the water column. It was assumed that the ratio between sediment and water column levels was the same as that currently measured on the Palos Verdes Shelf. The simulation for p,p'DDE was used to represent total DDTs, because p,p'DDE is on average greater than 90 percent of the total DDTs in fish from the Southern California Bight, based on analysis of all fish data in the HydroQual database.

For example, to produce a p,p'DDE level of 0.1 ppm wet-weight fillet in the white croaker in the vicinity of the Whites Point outfall (HydroQual segment 7), the required sediment

level is calculated to be 0.038 ppm dry (Table 1-2). To produce a PCB level of 0.1 ppm wet-weight fillet, the required sediment level is 0.083 ppm dry. It should be noted that to estimate levels in the sediment, a fraction organic carbon of 3.5 percent was used. These calculations can be adjusted for other values if necessary. To produce the same level in the kelp bass, the required water column levels are 1.1×10^{-7} ppm p,p'DDE and 6.7×10^{-7} ppm PCB. Results for the Dover sole are also given in Table 1-2. These results represent steady-state levels of p,p'DDE and PCBs that would produce the specified levels in these species of fish.

Table 1-2. Model Predictions for Critical Concentrations of Contaminants in Fish

Contaminant	Specified Level in Fish Fillets (ppm wet weight)	Species	Model-based exposure levels required to meet the specified fish levels:	
			Sediment (ppm dry)	Water Column (ppm dissolved)
p,p'DDE	0.1	White croaker	0.023	-
		Dover sole	0.22	-
		Kelp bass	-	8.9×10^{-8}
Total PCBs	0.1	White Croaker	0.036	-
		Dover Sole	0.18	-
		Kelp Bass	-	4.6×10^{-7}

Projection of future contaminant levels in the species of interest. In the model simulations performed for the fish in this study, temporally and spatially variable contaminant levels in the sediment and the water column were used to compute temporally and spatially variable fish levels. Similar calculations could be performed given projected future sediment and water column contaminant levels. The model would then project whole-body concentrations. Because quantitative relationships in contaminant levels exist among tissues, whole-body concentrations could be converted to, for example, equivalent fillet or gonad concentrations. In this way the model results could be restated in units that are relevant to regulatory or toxicological endpoints.

SECTION 2

DATA ANALYSIS

2.1 HYDROQUAL DATABASE

HydroQual has compiled an extensive database of contaminant data for the Southern California Bight (Table 2-1)[3]. This involved communication with many investigators, receipt of the data in electronic form or hard copy, and manipulation of the data from all sources to a common format. The current database contains data for several fish species, mussels and sediments spanning 1970 to 1995, including 7828 records. The database exists in a compressed format on a Hewlett Packard 800 series computer and is easily accessed by the HydroQual data analysis/graphics program. In addition, it can be exported to any platform as an ASCII file, or in spreadsheet format. Other data bases at HydroQual include data for sea lion and bird contaminant levels in the Southern California Bight. To our knowledge, this is the most complete collection of data from the Southern California Bight that exists in one location in a readily accessible form. The analyses presented below were based on statistics calculated from this database.

The database consists of samples collected and analyzed in previous studies, except for the data collected as part of the Southern California Bight Damage Assessment. This represents both a strength and a limitation of the analyses presented here. Conclusions based on observed spatial and temporal trends are strengthened to the extent that these patterns have been observed in multiple studies. On the other hand, analytical and sampling differences among studies make patterns more difficult to observe. Therefore, graphical comparisons among various studies were performed as part of this effort and are presented in Section 2.

The symbols on all data plots in Sections 2 and 3 represent arithmetic averages. The arithmetic mean was used, because the bioaccumulation models are based upon the principle of mass balance; that is, the models use the masses of contaminant in sediments and water to compute the mass of contaminant in the biota. The average mass in each compartment is directly related to the total mass (average = total ÷ number of observations). In contrast, other measures of location such as the median are not a simple function of the total mass.

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[3]Table 2-1 and Appendix D include only those records for data collected in the Southern California Bight (Mexican border to Point Conception). The database includes a considerable number of additional records of data collected in Mexico and north of the Bight.

Table 2-1. A Summary of the Data Included in the HydroQual Southern California DDT and PCB Database

Agency	Fish		Mussels		Sediment	
	No. Records	Years	No. Records	Years	No. Records	Years
California Mussel Watch Program			373	77-95		
Hyperion Treatment Plant	310	87-90				
LA County Sanitation District	1927	70-95			647	73-94
NOAA Benthic Surveillance					237	84-95
NOAA Mussel Watch	716	84-88	786	86-95	491	86-91
Pollock et al (1991)	996	87				
Santa Monica Bay Restoration Project	71	90				
Costa et al (1994)	146	92			213	91
Southern California Coastal Water Research Project Pilot Study					293	94-95
Garcelon et al (1994a,b)	32	92-93				
Martin et al (1984)			6	83		
Young (1982)			2	74		
Young et al (1982)			11	71-81		
Young and Heesen (1978)			17	71		
Young et al (1978)	358	73-79				
Risebrough (1987)	131	85	44	71-86	21	85-87
Overall	4,687	70-95	1,239	71-95	1,902	73-95

A measure of the uncertainty in the average concentration is useful for exploring trends and comparing model results with data. The standard error of the mean (=the standard deviation ÷ the square root of the number of observations) can be used to compute one measure of uncertainty. The vertical bars on all data plots in Sections 2 and 3 represent a spread of +/- two standard errors of the mean, unless otherwise noted.

An alternative method of data presentation is the Tukey box plot, in which the central tendency of the data are represented by the median and the actual spread of the data values is indicated by boxes, whiskers and symbols. Box plots are valuable because outliers are indicated individually and because no assumptions are made concerning the shape of the distribution of the data. In the body of the report, means +/- two standard errors are used, because means are required for model/data comparison, and because box plots sometimes result in cluttered figures, making it more difficult to interpret plots. However, because no single method of presentation is perfect, we provide Tukey box plots of all figures from Sections 2 and 3 in Appendices B and C, respectively.

2.2   NORMALIZATION OF SEDIMENT AND BIOTA DATA

The concentrations of p,p'DDE and PCBs in sediments from the study area have been generally reported as mass of contaminant per mass of dry sediment (i.e., ppm dry). For such hydrophobic contaminants, nearly all of the sorbed contaminant is associated with the particulate organic matter (POM) component of the sediment (USEPA 1993). Thus, mass of contaminant per mass of organic matter (or organic carbon) is a more meaningful measure of concentration that tends to reduce data variability. Additionally, this expression of concentration facilitates estimation of the dietary contaminant dose received by deposit-feeding animals. These animals derive their carbon and energy from the POM, and ingestion is expressed as mass organic carbon ingested per mass of animal per day. Therefore, for those sediment samples analyzed for organic carbon in addition to p,p'DDE and/or PCBs we have reexpressed concentrations as ppm OC (i.e., μg/g OC).

Lipids are generally acknowledged as the component of animal tissue where hydrophobic organic contaminants are stored. In an attempt to eliminate the fraction of variability amongst individual measurements due to differences in lipid content, p,p'DDE and PCB concentrations have been expressed as ppm lipid (i.e., μg/g lipid) where possible. Because lipid was measured

in a limited subset of the biota data, wet weight-based concentrations (i.e., ppm wet) were also used in many of the data analyses.

2.3 DEVELOPMENT OF A SEGMENT SCHEME FOR THE SOUTHERN CALIFORNIA BIGHT

To study spatial and temporal patterns in contaminant levels in sediment, water and biota, the Southern California Bight was divided into 15 segments. A visual examination of the sediment data was used to determine segment boundaries (see Figures 2-1 and 2-2). Boundaries between segments were chosen to provide spatial resolution sufficient for analysis of along-shore concentration gradients while providing data sufficient to compute regional average exposure concentrations for the biota. The locations of the individual sampling sites off the Palos Verdes shelf are presented in Figure 2-3. Although concentration differences exist between nearshore and offshore stations within segment transects, fish sampling locations are generally interspersed along sediment transects, and segment averages of data from all stations best represent surface sediment exposure concentrations. Therefore, spatial patterns were studied, by plotting the average concentration in each segment against distance from the center of that segment to the Los Angeles County outfall in segment 8. The segmentation schemes for the Southern California Bight and Palos Verdes Shelf (segments 3 through 10) regions are presented in Figures 2-4 and 2-5, respectively. Table 2-2 contains approximate segment boundary locations and names.

2.4 SEDIMENT

Analyses of p,p'DDE and total PCB concentrations in surface sediments (sediments with sample depths less than or equal to 5 cm) indicate that data from several sources exhibit similar spatial profiles throughout the Bight (see Figures 2-1 and 2-2). The sediment data from all studies were combined, and averages were computed for each of the 15 segments indicated on Figures 2-4 and 2-5.

Mean sediment p,p'DDE concentrations (ppm OC) declined by about a factor of 3 to 8 between 1973 and 1985 (Figure 2-6). A decline is seen in every segment. The trend of PCBs
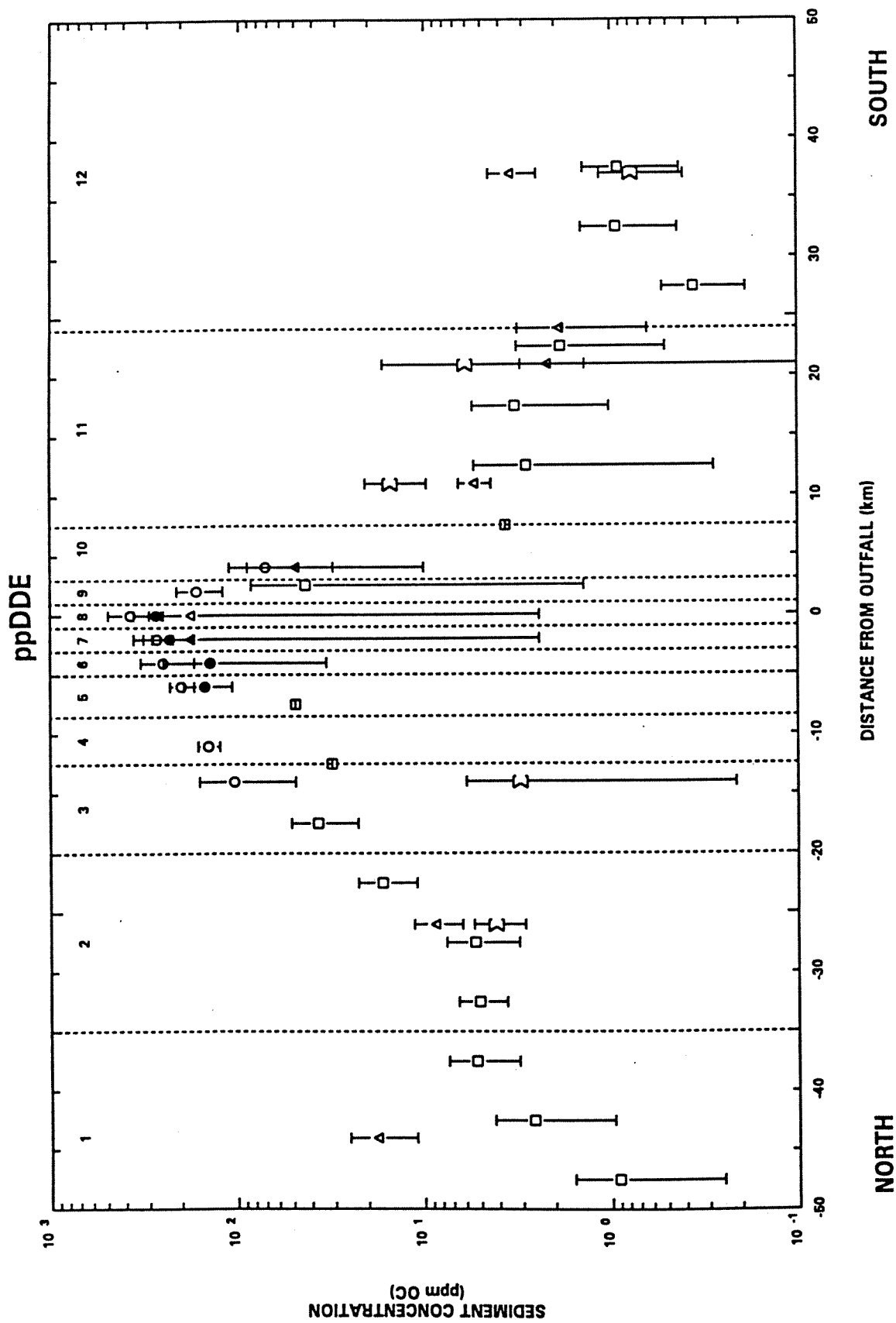
Figure 2-1. Observed p,p'DDE concentrations in surface sediments collected from the Southern California Bight between 1985 and 1995. Vertical dashed lines designate HydroQual segment boundaries. Segment numbers (1-12) are indicated at the top of the figure. Concentrations are expressed as ppm organic carbon (arithmetic means +/- 2 standard errors of the mean) and are plotted as a function of distance from the Los Angeles County Outfall (segment 8; kilometer 0). (Symbols: ○ - Los Angeles Sanitation District; Σ - NOAA Benthic Surveillance; Δ - NOAA Mussel Watch Program; ● - SCBDA data; □ - Southern California Coastal Water Research Project Pilot Study).
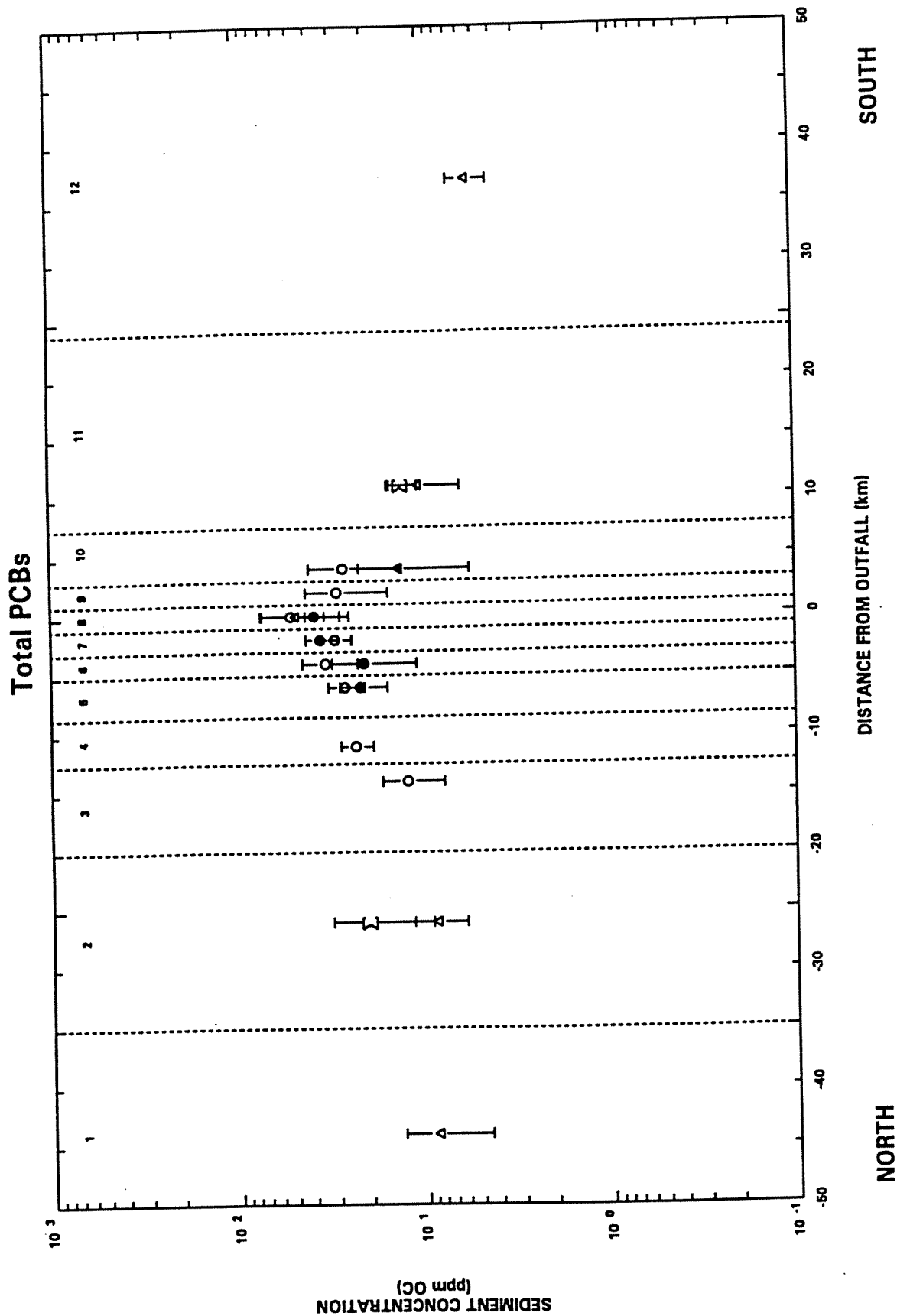
Total PCBs

Figure 2-2. Observed total PCB concentrations in surface sediments collected from the Southern California Bight between 1985 and 1995. Vertical dashed lines designate HydroQual segment boundaries. Segment numbers (1-12) are indicated at the top of the figure. Concentrations are expressed as ppm organic carbon (arithmetic means +/- 2 standard errors of the mean) and are plotted as a function of distance from the Los Angeles County Outfall (segment 8; kilometer 0). (Symbols: O - Los Angeles Sanitation District; ● - SCBDA data; Σ - NOAA Benthic Surveillance; Δ - NOAA Mussel Watch Program).
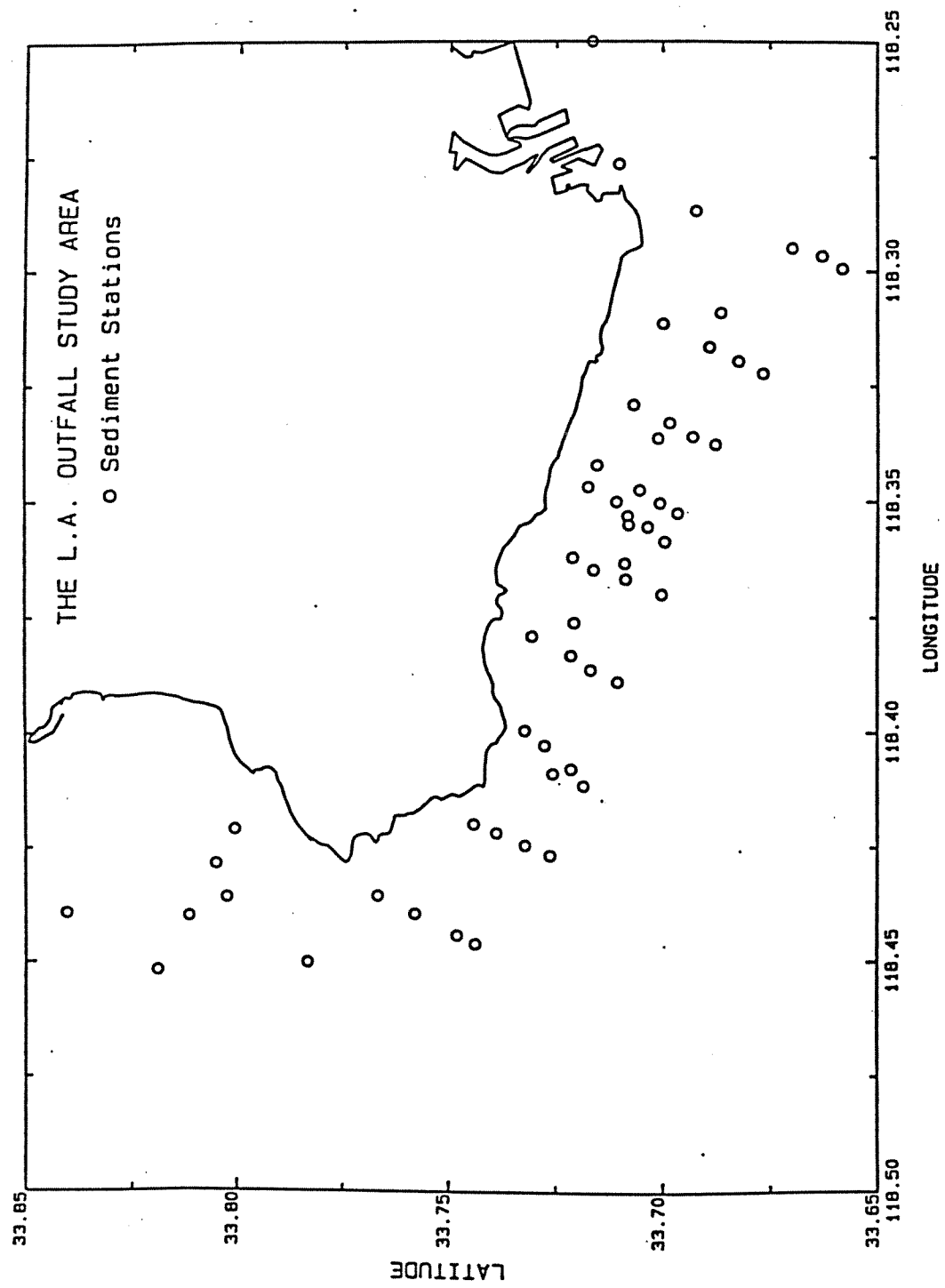
Figure 2-3. Sediment sampling station locations in Palos Verdes study area.
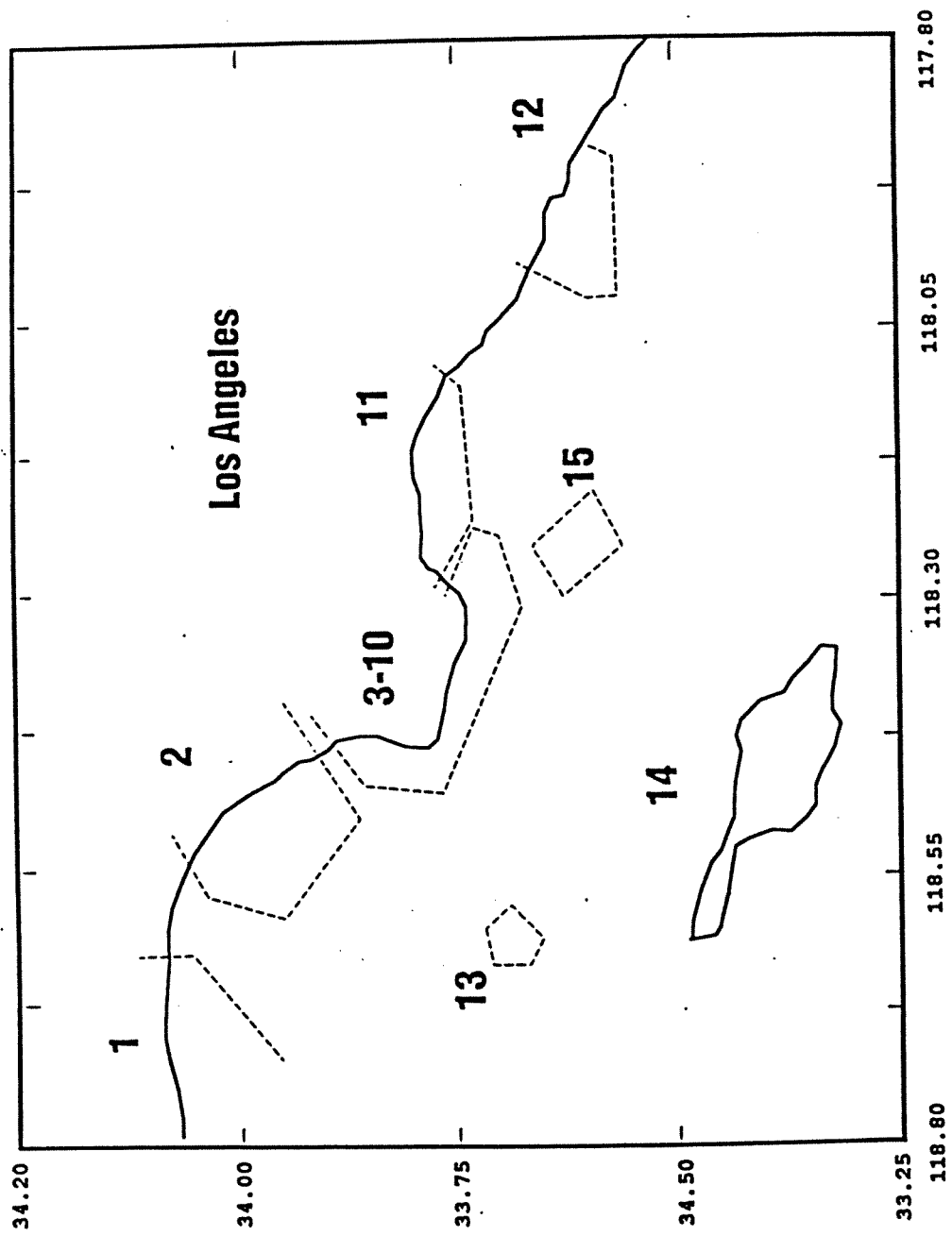
Figure 2-4. Segmenting scheme developed for analysis of DDE and PCB data collected in the Southern California Bight
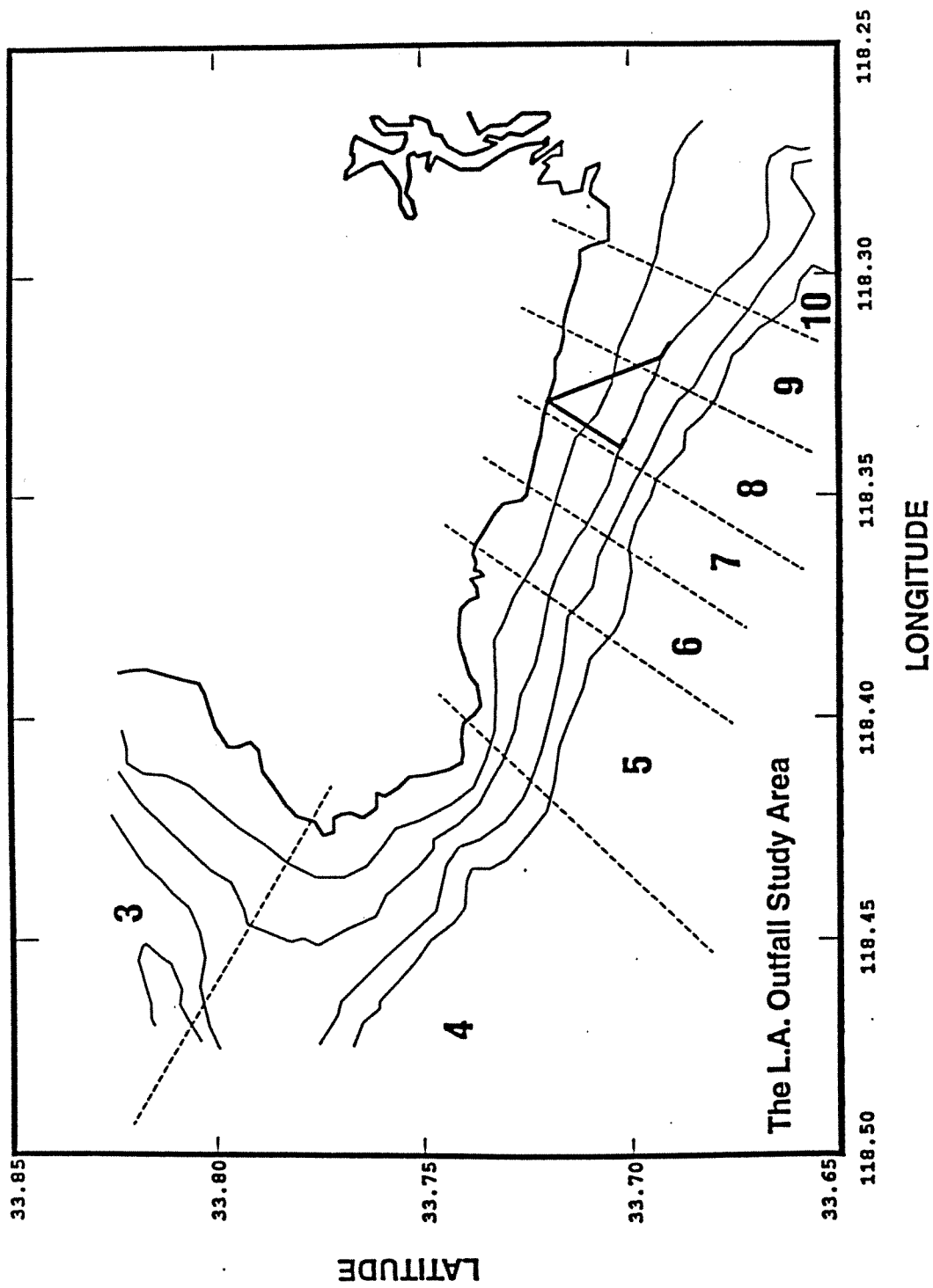
Figure 2-5. Segmenting scheme developed for analysis of DDE and PCB data collected in the Palos Verdes region

Table 2-2. Description of the Segments into which the Study Area of the Southern California Bight was Divided for Purposes of Data Analysis

Approximate Segment Boundaries	HQI Segment Number	LA Transects Included in Segment	HQI Segment Name	Distance in Kilometers from LA Outfall*
Point Dume to Santa Monica	1		Point Dume - SMB	-44.0
Santa Monica to Hermosa Beach	2		Upper Santa Monica Bay	-26.0
Hermosa Beach to Rocky Point	3	0	Lower Santa Monica Bay	-14.0
Rocky Point to Long Point	4	1,2	Palos Verdes Point	-11.0
Long Point to Portuguese Point	5	3,4	Western PV Shelf	-6.0
Portuguese Point to Bunker Point	6	5	West-Central PV Shelf	-4.0
Bunker Point to KOU Reef	7	6,7	East-Central PV Shelf	-2.0
KOU Reef to Royal Palms	8	8	West Outfall	0.0
Royal Palms to Point Fermin	9	9	East Outfall	2.0
Point Fermin to Cabrillo Beach Park	10	10	PVS/LAH	4.0
Cabrillo Beach Park to Long Beach	11		LA Harbor	11.0
Vicinity of Newport Beach	12		Newport Beach	37.0
Northwest of Santa Catalina Island	13		S. Catalina NW Offshore	-20.0
Santa Catalina Island	14		Santa Catalina Island	35.0
PVS/LAH Offshore	15		PVS/LAH Offshore	14.0
Newport Beach to San Diego	21		South of Newport Beach	100.0
Santa Cruz Island	22		Santa Cruz Island	-160.0
San Miguel Island	23		San Miguel Island	-160.0
San Clemente Island	28		San Clemente Island	92.0
Santa Barbara Island	51		Santa Barbara Island	-72.0
Anacapa Island	52		Anacapa Island	-108.0
Santa Rosa Island	53		Santa Rosa Island	-160.0
Western San Pedro Channel	60		San Pedro Channel A	-44.0
West-Central San Pedro Channel	61		San Pedro Channel B	-32.0
East-Central San Pedro Channel	62		San Pedro Channel C	22.0
Eastern San Pedro Channel	63		San Pedro Channel D	44.0

*Note: Distances north of the LA County outfall are negative.
Distances south of the LA County outfall are positive.
These distances can be used to interpret the spatial plots of Section 2.
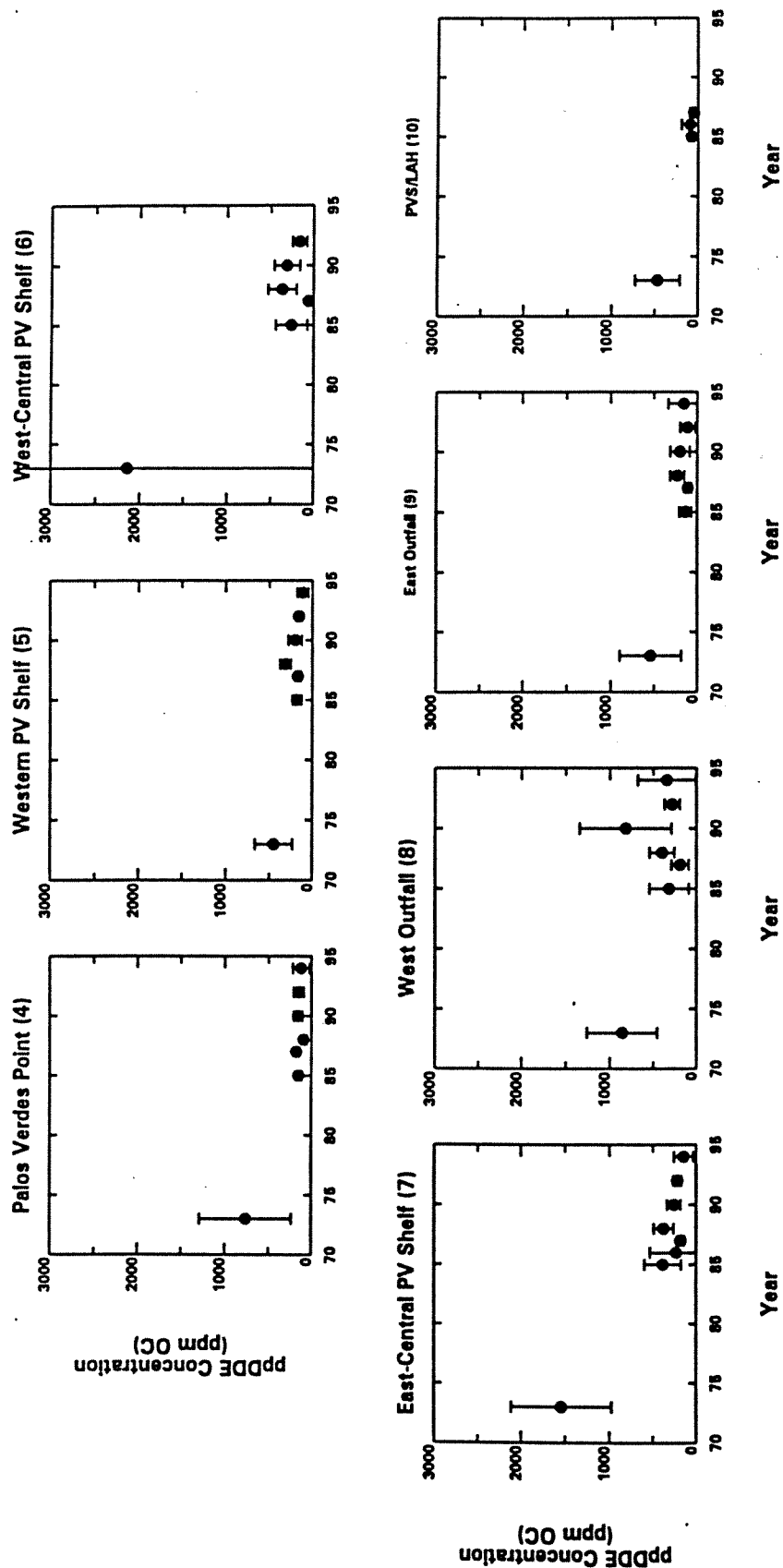
Figure 2-6.   Temporal profiles of p,p'DDE concentrations in surface sediments for various segments of the Palos Verdes Shelf (ppm organic carbon; arithmetic mean +/- 2 standard errors of the mean).

2-12

over the same period cannot be ascertained because of a lack of data in the 1970's (Figure 2-7). Since 1985, the concentrations of both p,p'DDE and PCB have varied between years with no observable trend.

Between 1972 and 1980, loads of total DDT from the outfall declined more than 10-fold (Young et al. 1988). The observations that surface sediment concentrations declined less rapidly than loadings in the 1970's, and that sediment concentrations in the absence of significant loadings exhibit no apparent trend (1985 through 1995) indicate a low sediment burial rate and a relatively long half-life.

Spatial patterns were evaluated for the period 1985 through 1995, a period for which no apparent temporal trends exist. Concentration values for each modeling segment were averaged over the period and the averages are presented as a function of distance from the outfall in Figure 2-8. South of the outfall, organic carbon normalized p,p'DDE levels in the sediments decline by a factor of about 35 (from 350 ppm to 10 ppm) within the first 10 kilometers. North of the outfall, p,p'DDE levels decline less dramatically, by a factor of 2 to 3, within 10 kilometers of the outfall. PCB levels in sediments are about 10-fold lower than p,p'DDE levels. They decrease in a manner similar to p,p'DDE. Both decline at a rate of about 10 percent per kilometer for the first 14 km to the north. Note that dry weight concentrations decline more rapidly than the carbon weight concentrations, because the organic content of the sediments also declines away from the outfall.

The more gradual decline to the north probably reflects more extensive transport in this direction with the dominant northward-flowing average bottom current over the Palos Verdes shelf.

2.5   WATER

There have been few direct measurements of water column concentrations of p,p'DDE and PCB in the Southern California Bight. However, water column levels can be inferred from concentrations in mussels, and a considerable database exists for levels of p,p'DDE and PCB in mussels. A steady-state bioaccumulation factor (BAF) is necessary to convert from mussel to water. Several mussel BAF values have been reported from measurements of p,p'DDE concentrations in mussel and water samples collected from field sites in San Francisco Bay (Lee et al. 1994). Lipid-based values based on unfiltered water measurements averaged 8.9×10^6
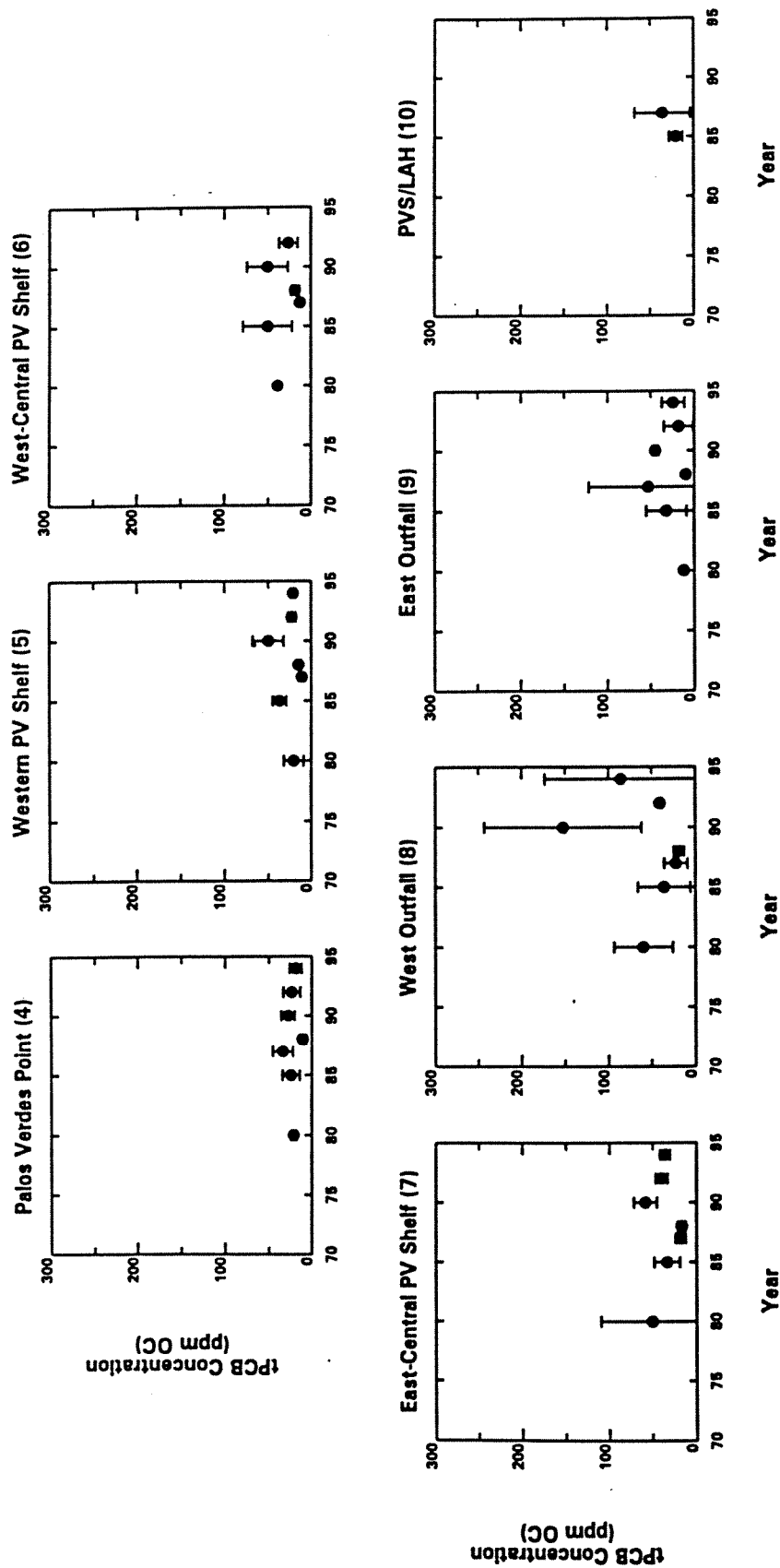
Figure 2-7.   Temporal profiles of total PCB concentrations in surface sediments for various segments of the Palos Verdes Shelf (ppm organic carbon; arithmetic mean +/- 2 standard errors of the mean).
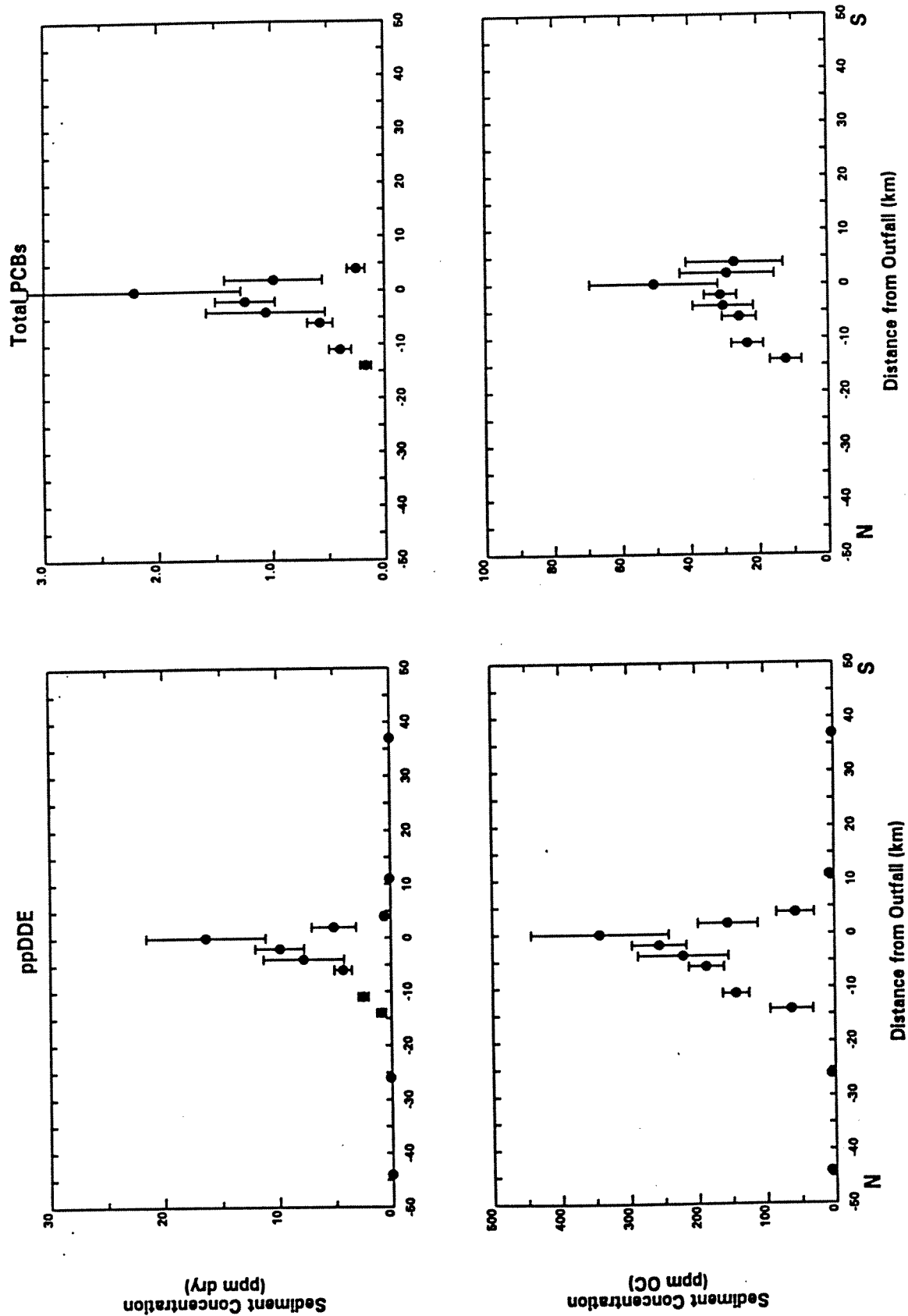
Figure 2-8. Spatial profiles of p,p'DDE and Total PCB levels in surface sediments collected from the Southern California Bight between 1985 and 1995. Top panels show data expressed as ppm dry weight. Bottom panels show data expressed as ppm organic carbon; (arithmetic mean +/- 2 standard errors of the mean). Zero km point at Whites Point Outfall. Distances to the north are negative.

liters/kg lipid (range 3 to 20 X 10^6). This value was used to estimate water column p,p'DDE concentrations in the Southern California Bight. This value is similar to the measured value of the octanol/water partition coefficient (K_{ow}) for p,p'DDE (9.1 X 10^6, de Bruijn et al. 1989). Therefore, K_{ow} is a measure of BAF. The use of K_{ow} as an estimate of the mussel BAF is supported by paired field measurements of organochlorine concentrations in caged mussels and in the dissolved phase in New Bedford Harbor (Hofelt & Shea 1997). K_{ow} for Total PCBs was used to estimate the lipid-based BAF value for PCBs.

The K_{ow} value for total PCBs can be estimated from knowledge of the average K_{ow} value for each homolog and the homolog distribution in the mussels of the Bight. Based on values of K_{ow} measured for each of the PCB congeners, approximate average log (K_{ow}) values for the homologs are: 5.0 (di), 5.5 (tri), 6.0 (tetra), 6.5 (penta), 7.0 (hexa), 7.5 (hepta), 8.0 (octa), and 8.5 (nona) (Hawker and Connell 1988). The log (K_{ow}) values estimated by Hawker and Connell (1988) are on average 0.3 log units lower than the estimates of de Bruijn et al. (1989). Because the log (K_{ow}) value of de Bruijn et al. (1989) was used for p,p'DDE, the values of Hawker and Connell (1988) were adjusted by adding 0.3.

The NOAA Mussel Watch dataset included within the HydroQual Database contains concentrations for each of the 10 homologs in mussels collected in 1986 in several regions of the Bight. The average of the homolog-specific K_{ow} values, weighted by their proportion in the Southern California Bight mussels, yields an average log (K_{ow}) value of 6.3 for total PCBs in mussels. This value was used to estimate dissolved water column total PCB concentrations from concentrations in mussels.

Computed water column contaminant levels are directly proportional to levels in mussels. Therefore, spatial and temporal trends observed in mussels are presented as surrogates for the trends in water. Mussel and sediment spatial profiles for 1985-1995 are compared in Figures 2-9 (p,p'DDE) and 2-10 (PCB). Mussel p,p'DDE concentrations are highest near the outfall and decline by a factor of 3 to 5 within 25 km to the north of the outfall. In contrast, p,p'DDE concentrations in the sediments decline by up to 50-fold within the same distance. This pattern difference probably reflects differences in the transport of dissolved and particulate p,p'DDE fractions. Mussel PCB levels are more variable spatially than p,p'DDE levels, with highest values observed away from the outfall in Los Angeles Harbor (km 11) and in upper Santa Monica Bay in the vicinity of the Hyperion Outfall (km -26) (Figure 2-10). In contrast, PCB concentrations in sediment decline monotonically away from the outfall.
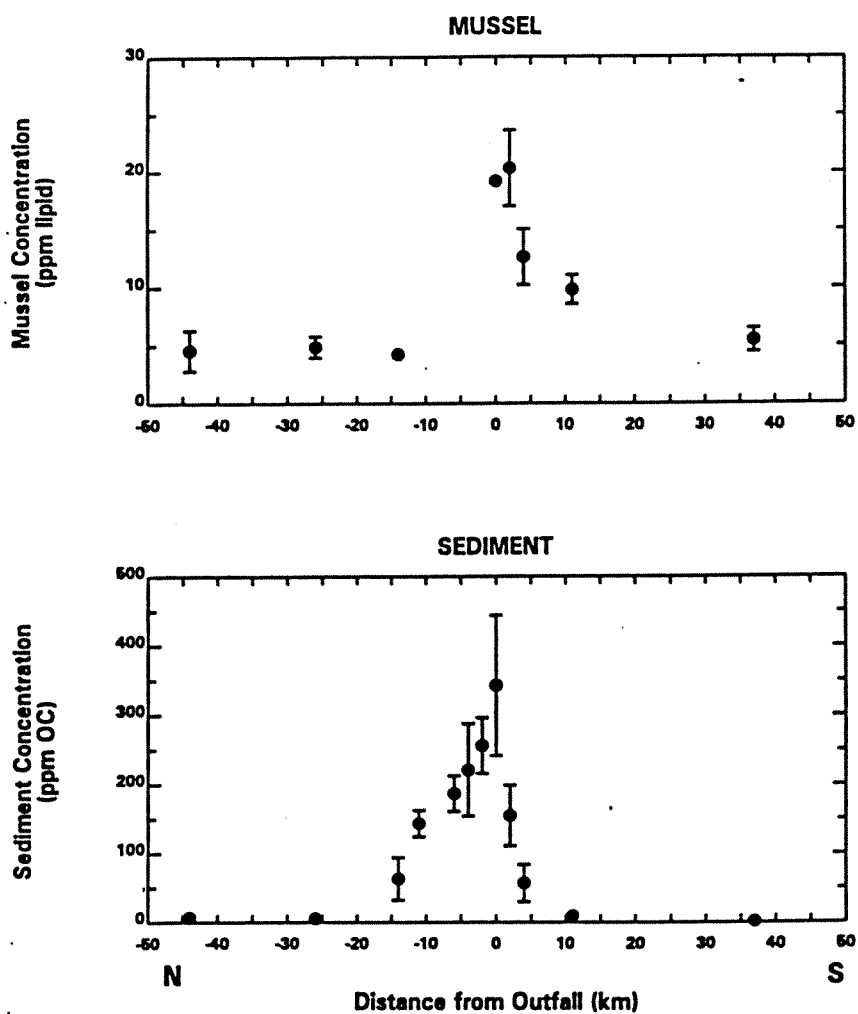
Figure 2-9. Spatial profiles of p,p'DDE concentrations in mussels and surface sediments
collected from the Southern California Bight between 1985 and 1995. (ppm
lipid, organic carbon; arithmetic mean +/- 2 standard errors of the mean).
Zero km point at Whites Point Outfall. Distances to the north are negative.

MUSSEL

SEDIMENT

N   S

Distance from Outfall (km)
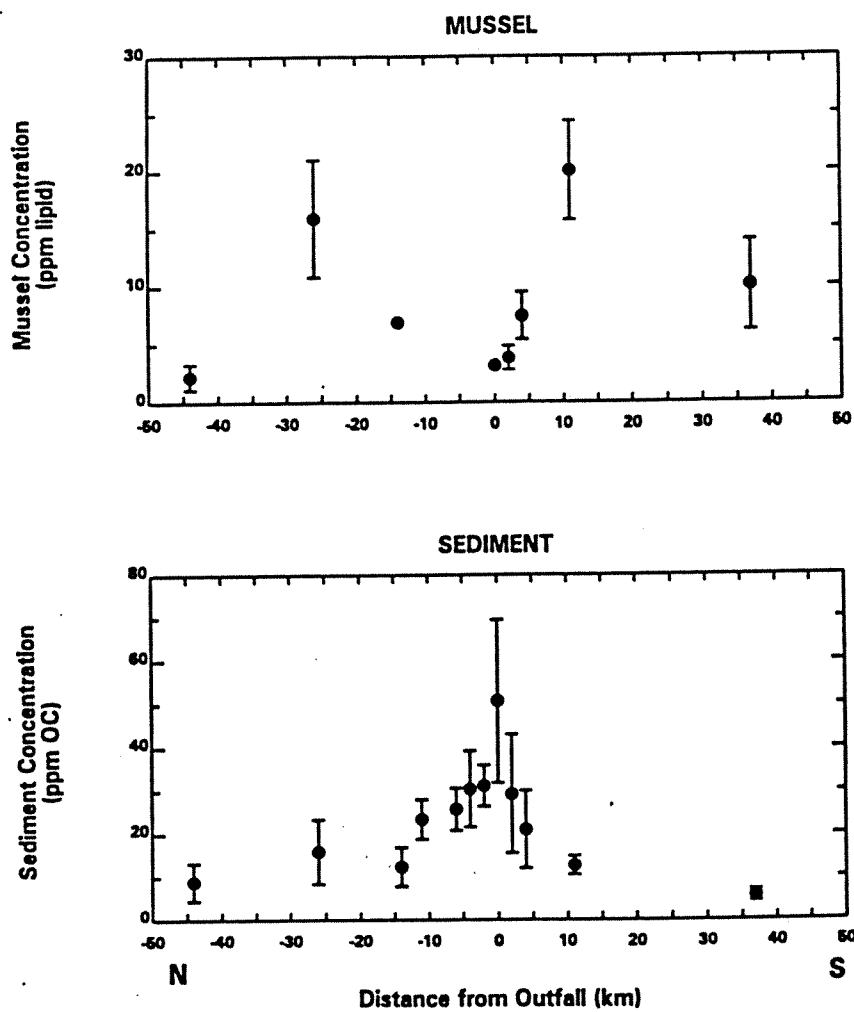
Figure 2-10. Spatial profiles of total PCB concentrations in mussels and surface sediments collected from the Southern California Bight between 1985 and 1995. (ppm lipid, organic carbon; arithmetic mean +/- 2 standard errors of the mean). Zero km point at Whites Point Outfall. Distances to the north are negative.

Continental scale (Mexico to Alaska) spatial profiles of mussel wet-weight p,p'DDE levels are presented in Figure 2-11[4]. Up to the mid-1980's the peak at the outfall exceeds all other areas by one to two orders of magnitude. In the late 1980's a peak at the outfall is still evident but the average concentration in California north of Point Conception (-250 to -700 km) is now approximately one-half the concentration near the outfall. This change from the previous periods occurs because of differences in the rates at which concentrations in the two areas have declined. Peak concentrations in the vicinity of the outfall decline by about 98 percent (ca. 4 to 0.1 ppm wet), whereas concentrations in northern California decline by about 50 percent (ca. 0.1 to 0.06 ppm wet). Further analyses of the data included in northern California (not shown) indicate that elevated concentrations occur mainly in two locations: Monterey Bay and San Francisco Bay (-350 and -500 km, respectively). Even in these locations, values are approximately 4-fold lower than the values at the outfall.

Of particular significance is the data for 1980 through 1984 which includes samples from along the Mexican coast, an area sometimes mentioned as a possible region of high contamination. The Palos Verdes shelf concentration peak during that period is an order of magnitude higher than the highest measurement from Mexico and two orders of magnitude higher than the other seven Mexican locations (Figure 2-11).

Temporal profiles of mussel p,p'DDE concentrations suggest that nearly all of the decline from early 1970's levels occurred prior to the mid-1980's (Figure 2-12). As with sediment p,p'DDE concentrations, from about 1985 to 1995 concentrations off the Palos Verdes shelf (segments 8-11) and in Santa Monica Bay (segment 2) exhibit no consistent temporal trend.

2.6 FISH

Several fish studies were incorporated into our database, including the LACSD database, the Santa Monica Bay Restoration Project, Pollock (1991) and the Hyperion Treatment Plant study (see Table 2-1). The data from each of these studies indicate generally consistent spatial patterns in which the highest p,p'DDE concentrations occur in the vicinity of the Whites Point outfall; arithmetic mean values for each of three species in each HydroQual segment for each data source are presented in Figure 2-13. In subsequent analyses, fish fillet data from all sources (except as specifically noted) were combined.

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[4]In figure 2-11, each value presented between -250 and -750 km represents the average for a 100 km region of northern California, with the exception of the point at -500 km, which represents a 200 km region.
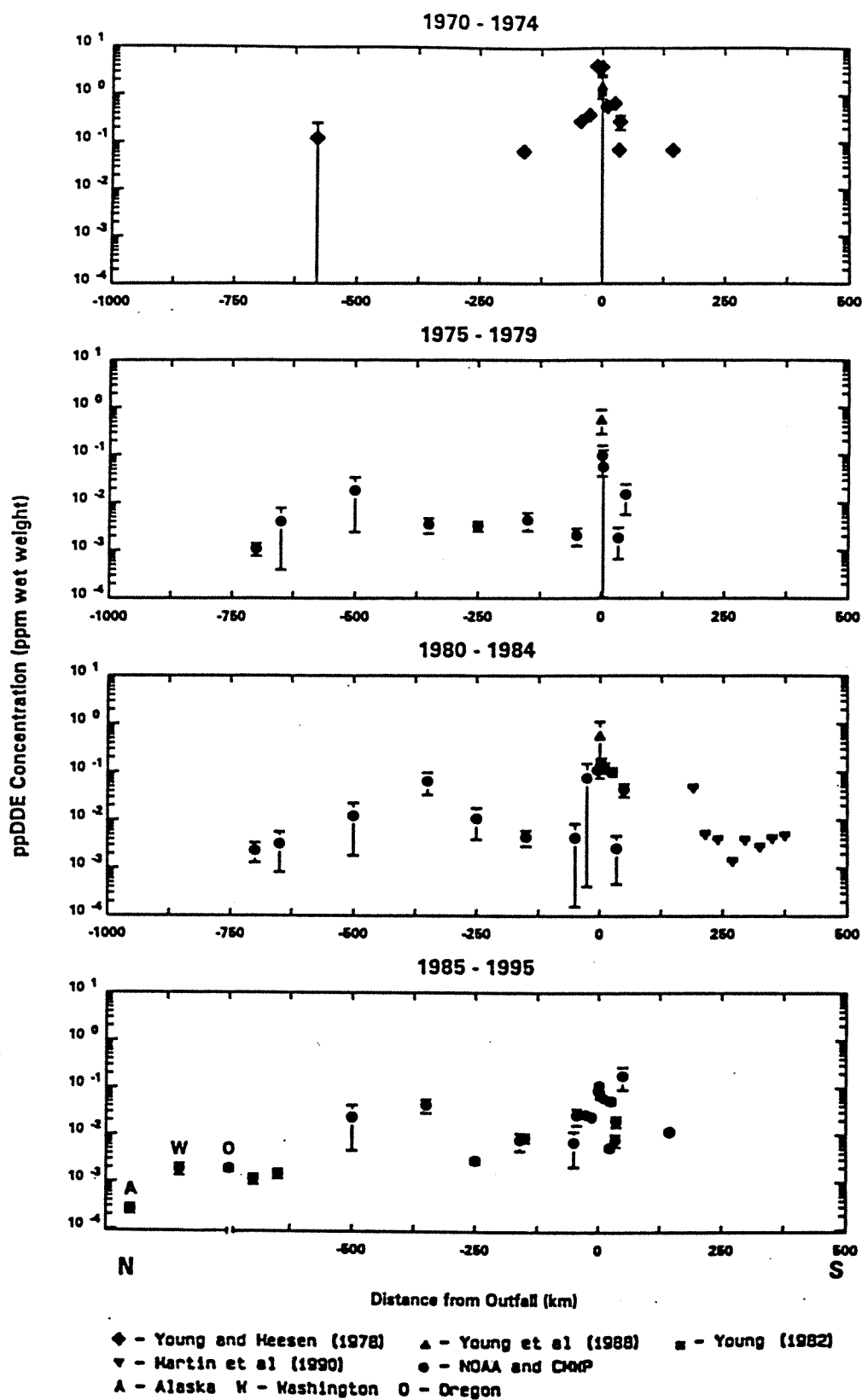
Figure 2-11. Spatial profiles of mussel p,p'DDE concentrations along the west coast between Alaska and Mexico for four time periods. (ppm wet weight; arithmetic mean +/- 2 standard errors of the mean). Zero km point at Whites Point Outfall. Distances to the north are negative.
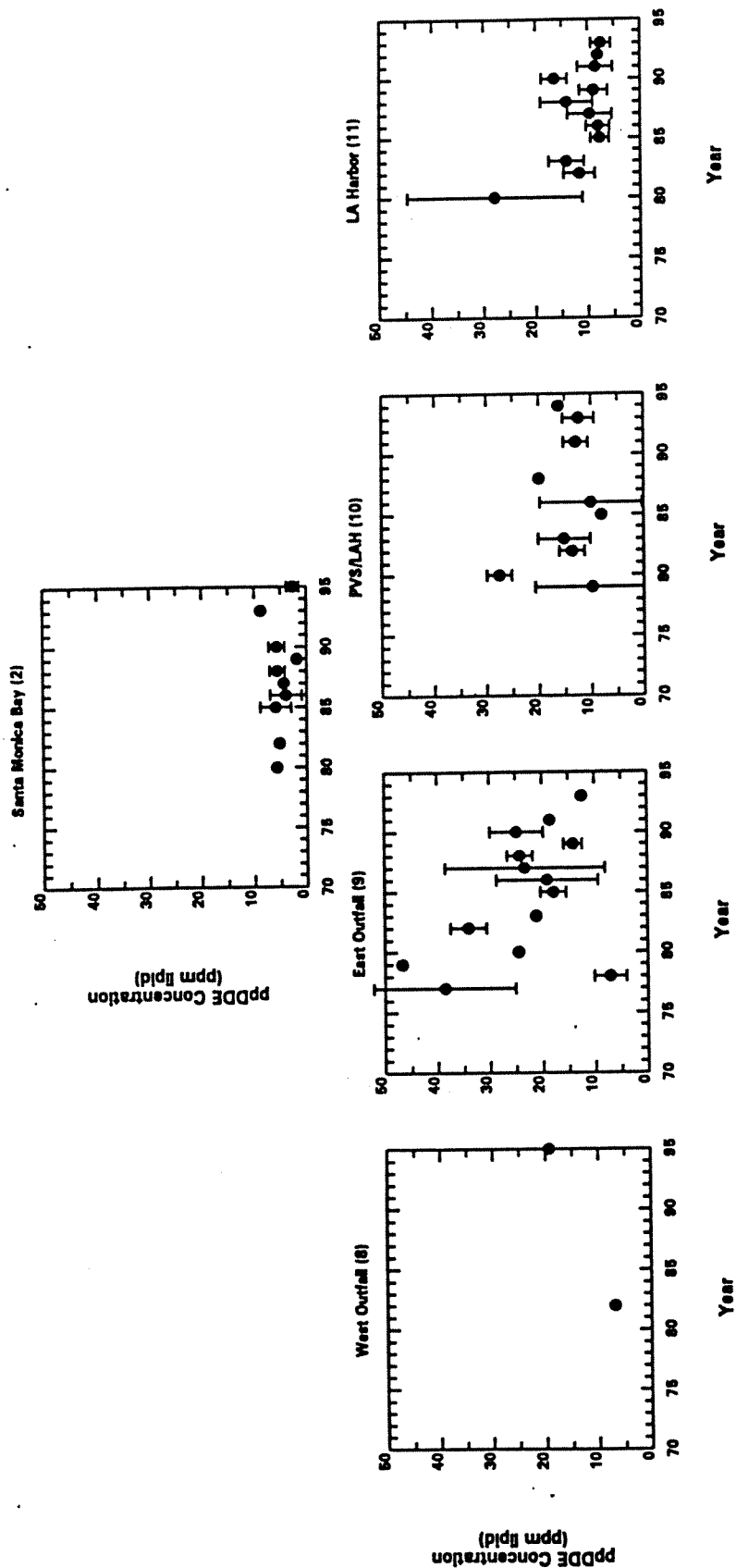
Figure 2-12. Temporal profiles of p,p'DDE concentrations in mussels for selected segments of the Southern California Bight. (ppm lipid; arithmetic mean +/- 2 standard errors of the mean).
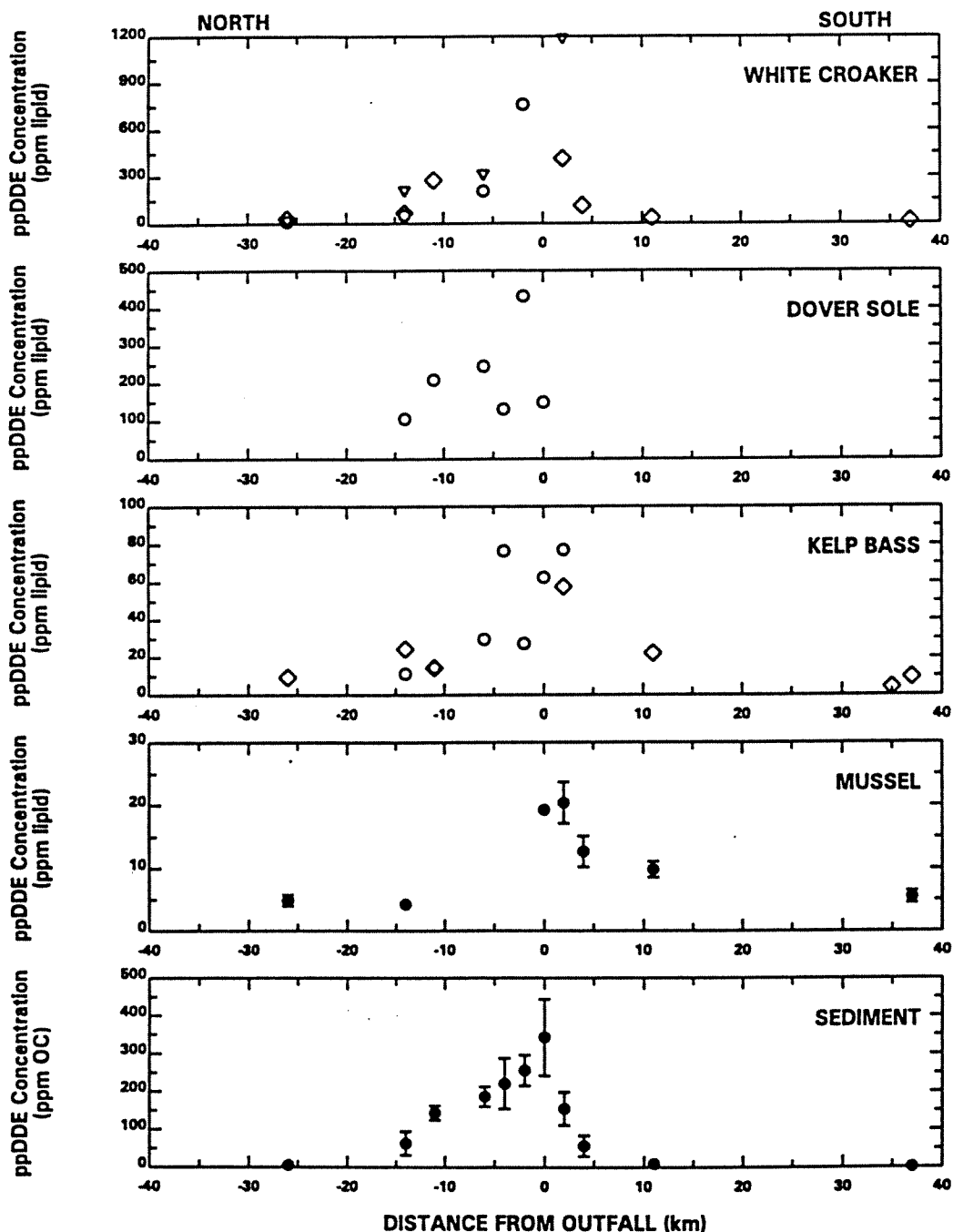
Figure 2-13. Observed p,p'DDE concentrations in fish, sediment and mussels collected from the Southern California Bight between 1985 and 1995. Concentrations in fish fillets and mussels are expressed as ppm lipid. Surficial sediment concentrations are expressed as ppm organic carbon (arithmetic means +/- 2 standard errors of the mean). All data are plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). (Symbols: O - Los Angeles Sanitation District; ◇ - Pollock et al, 1991; ▽ - Santa Monica Bay Restoration Project, 1992). Mussel and sediment data are presented as averages for each segment including all studies.

Spatial profiles of p,p'DDE levels in white croaker, Dover sole and kelp bass for the period 1985 to 1995 are presented in Figure 2-13 along with similar profiles for surface sediments and mussels. Croaker in the vicinity of the outfall have concentrations of about 1000 ppm lipid. At a distance of about 10 km from the outfall concentrations are substantially lower. To the south a reduction of more than a factor of 10 is seen. To the north the decline is less; about a factor of 3 to 4. Moving further out concentrations continue to decline. Overall this spatial pattern is similar to that observed in the sediment.

The kelp bass spatial profile of p,p'DDE concentrations also shows a peak in the vicinity of the outfall. However, concentrations in this area are about 60 to 70 ppm lipid; about a factor of 15-20 less than in croaker. This difference in extent of contamination probably reflects the pelagic nature of the kelp bass food web in contrast to the strong benthic association of the croaker food web. Differences in the contributions of water column and sediment p,p'DDE probably also account for the slower decline of kelp bass p,p'DDE away from the outfall, particularly to the south. This slower decline is consistent with the differences in decline between sediments and mussels.

In contrast to the white croaker and kelp bass, the Dover sole spatial profile of p,p'DDE concentrations has a distinctive peak at the outfall, but lacks evidence of a smooth spatial gradient with distance from the outfall. Lowest concentrations occur furthest from the outfall (mean of 100 ppm lipid 14 km to the north). Concentrations in the vicinity of the outfall average between 120 and 420 ppm lipid. The lower values are slightly above the values seen in kelp bass, whereas the highest values are about a factor of two below concentrations in white croaker. Because only juvenile Dover sole are found on the Palos Verdes Shelf, the data are reflective of animals that had spent one to two years feeding in the water column and a time feeding in the sediment that depended on their age at capture and age at settlement in the benthos (see Section 3.5.1.3). Thus, it makes sense that Dover sole contaminant concentrations are between those observed in kelp bass and white croaker. The variability of the data may also be related to the pelagic:benthic sequence, i.e., a consequence of differences in exposure time to the sediment and water column among the animals collected.

PCB levels in white croaker, Dover sole and kelp bass for the period 1985 to 1995 (Figure 2-14) exhibit spatial patterns similar to p,p'DDE, but with less decline away from the outfall and greater variability. Levels approximating those near the outfall were measured 10 to 15 km north of the outfall for all three species. These locations may be sites of local PCB
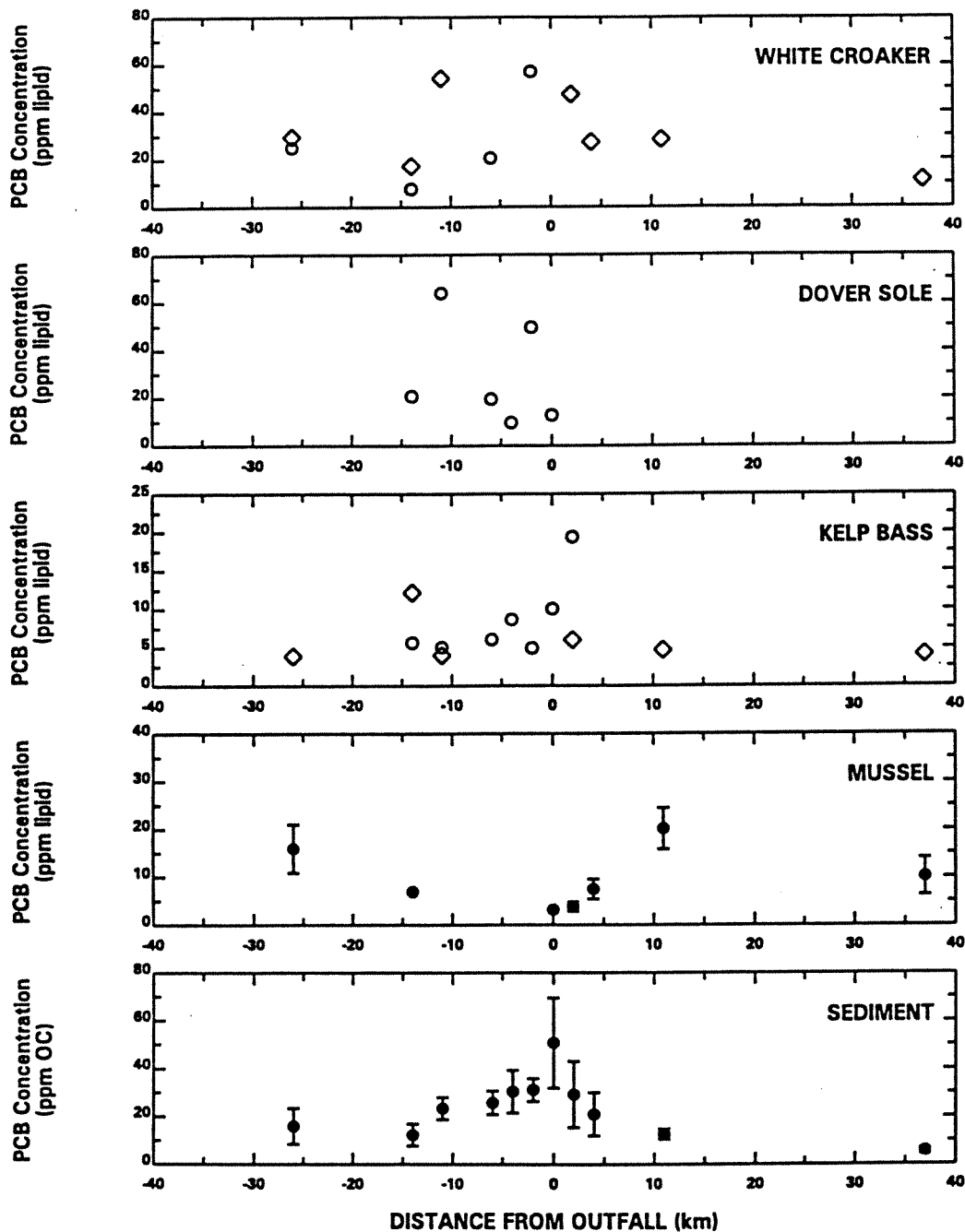
Figure 2-14. Observed total PCB concentrations in fish, sediment and mussels collected from the Southern California Bight between 1985 and 1995. Concentrations in fish fillets and mussels are expressed as ppm lipid. Surficial sediment concentrations are expressed as ppm organic carbon (arithmetic means +/- 2 standard errors of the mean). All data are plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). (Symbols: ○ - Los Angeles Sanitation District; ◇ - Pollock et al, 1991). Mussel and sediment data are presented as averages for each segment including all studies.

contamination, as was indicated in the mussel data.

Temporal trends in wet-weight-based contaminant concentrations were explored because the amount of lipid data was limited. p,p'DDE concentrations in kelp bass and Dover sole were variable in time. Declines in p,p'DDE levels from 1970 to the mid-1980's ranged from no decline near the LA outfall to declines by a factor of about 10 at Palos Verdes Point and lower Santa Monica Bay (Figures 2-15 and 2-16). PCBs exhibited similar temporal trends but a slight decline was observed for both fish near the outfall (Figures 2-17 and 2-18). Temporal trends from 1970 to the mid-1980's were not evaluated for white croaker due to a lack of sufficient p,p'DDE and PCB data (Figures 2-19 and 2-20). p,p'DDE and PCB concentrations in all three species have not exhibited a consistent downward trend since the mid-1980's.

2.7   SEA LIONS

Concentrations of p,p'DDE and PCB in the blubber of sea lions from San Miguel Island have been measured in three separate studies (Table 2-3). All of these studies focused on two groups of postpartum females: premature parturient and full-term parturient. The first study (DeLong et al. 1973) included 6 premature parturient and 4 full-term parturient females that were collected within 24 hours of parturition (April 20 to 24, 1970 for the premature and June 24 to 26, 1970 for the full-term). This study found that the premature parturient females had much higher p,p'DDE and PCB concentrations (two to eight times) than the full-term parturient females. These females were also much younger (mean age of 8 years in comparison to 12 years for the full-term parturient females). A follow-up study was conducted in 1972 (Gilmartin 1976). Ten premature parturient females were collected between March 27 and 29 and April 25 and 27. Ten full-term parturient females were collected between June 13 and 15. Again, the premature parturient group were much younger (6 to 8 years old versus 10 to 14 years old) and had much higher p,p'DDE and PCB concentrations (about 4 to 8 times). Both of these studies suggested that the higher contaminant concentrations may be responsible for the premature parturition, although Gilmartin and coworkers found a viral infection that may have also been a causative agent.
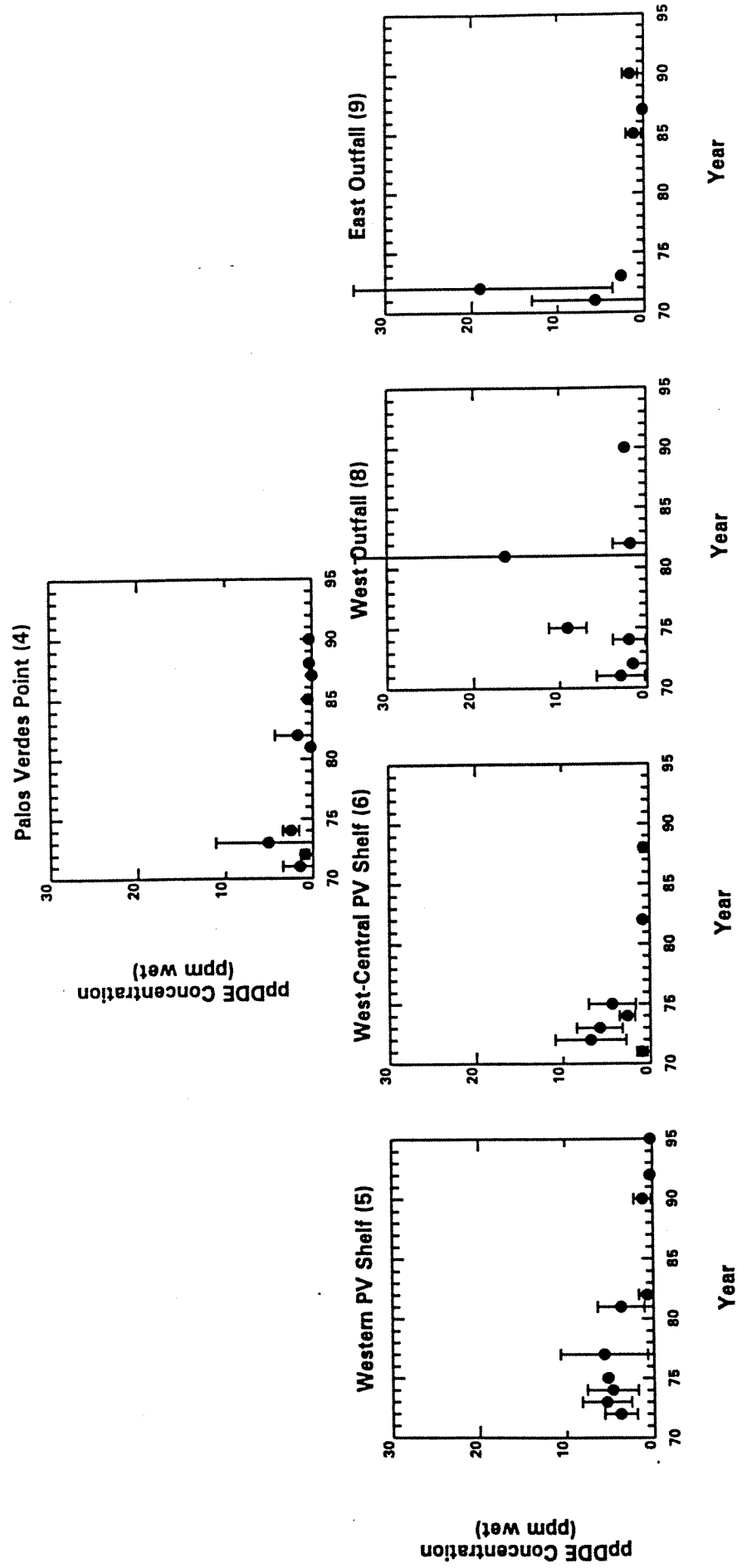
Figure 2-15. Temporal profiles of p,p'DDE concentrations in kelp bass for selected segments of the Southern California Bight. (ppm wet weight; +/- 2 standard errors of the mean).
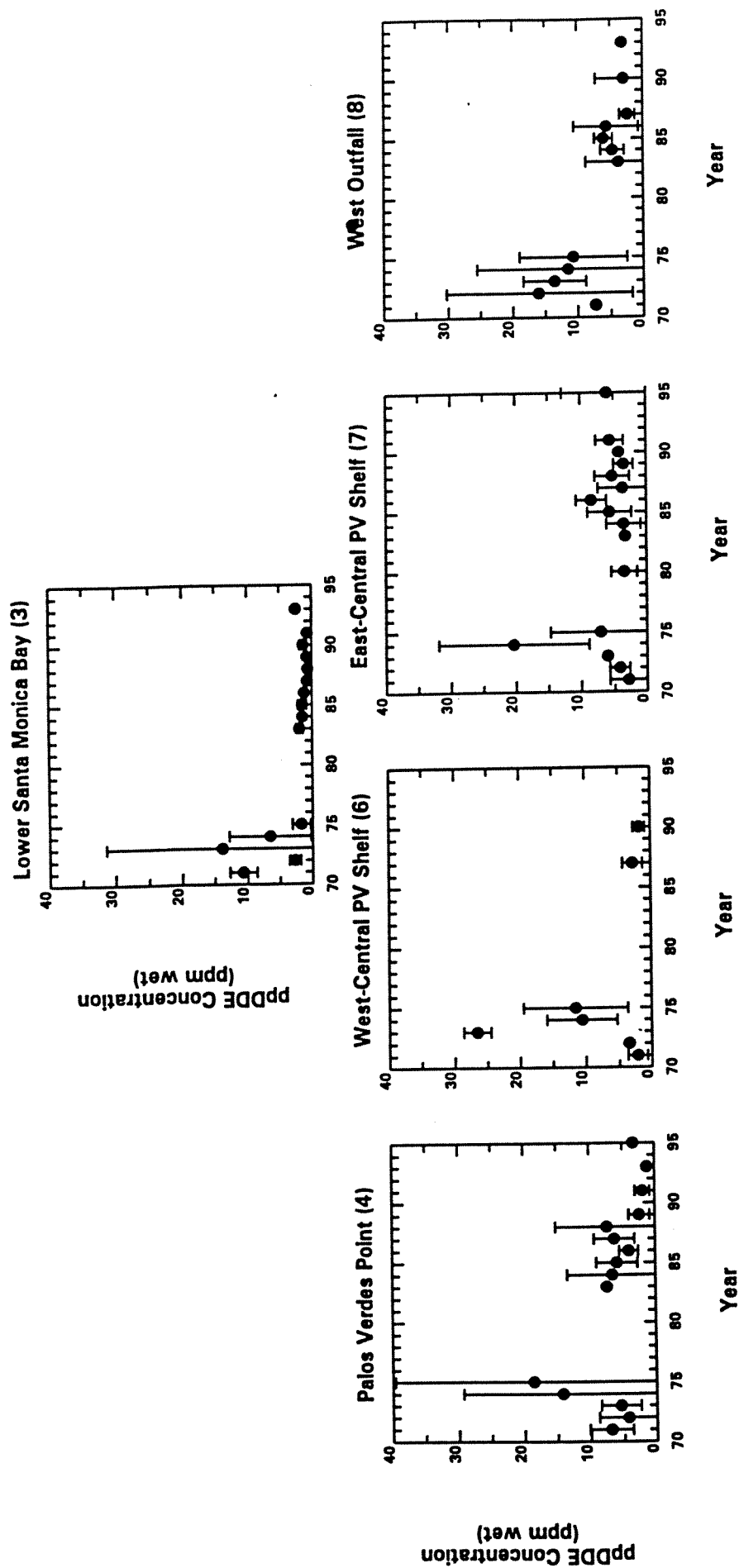
Figure 2-16. Temporal profiles of p,p'DDE concentrations in Dover sole for selected segments of the Southern California Bight (ppm wet weight; +/- 2 standard errors of the mean).
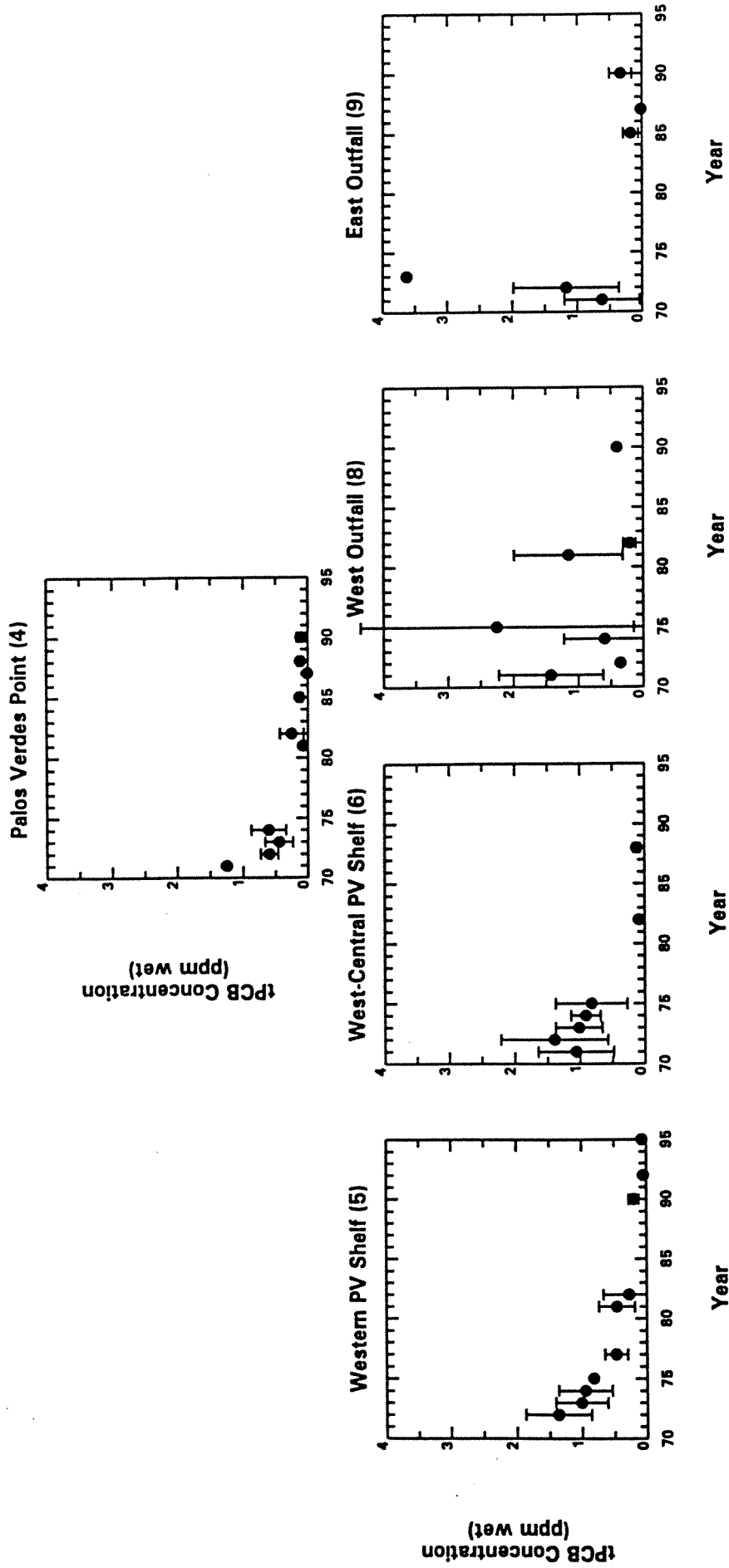
Figure 2-17. Temporal profiles of total PCB concentrations in kelp bass for selected segments of the Southern California Bight. (ppm wet weight; +/- 2 standard errors of the mean).
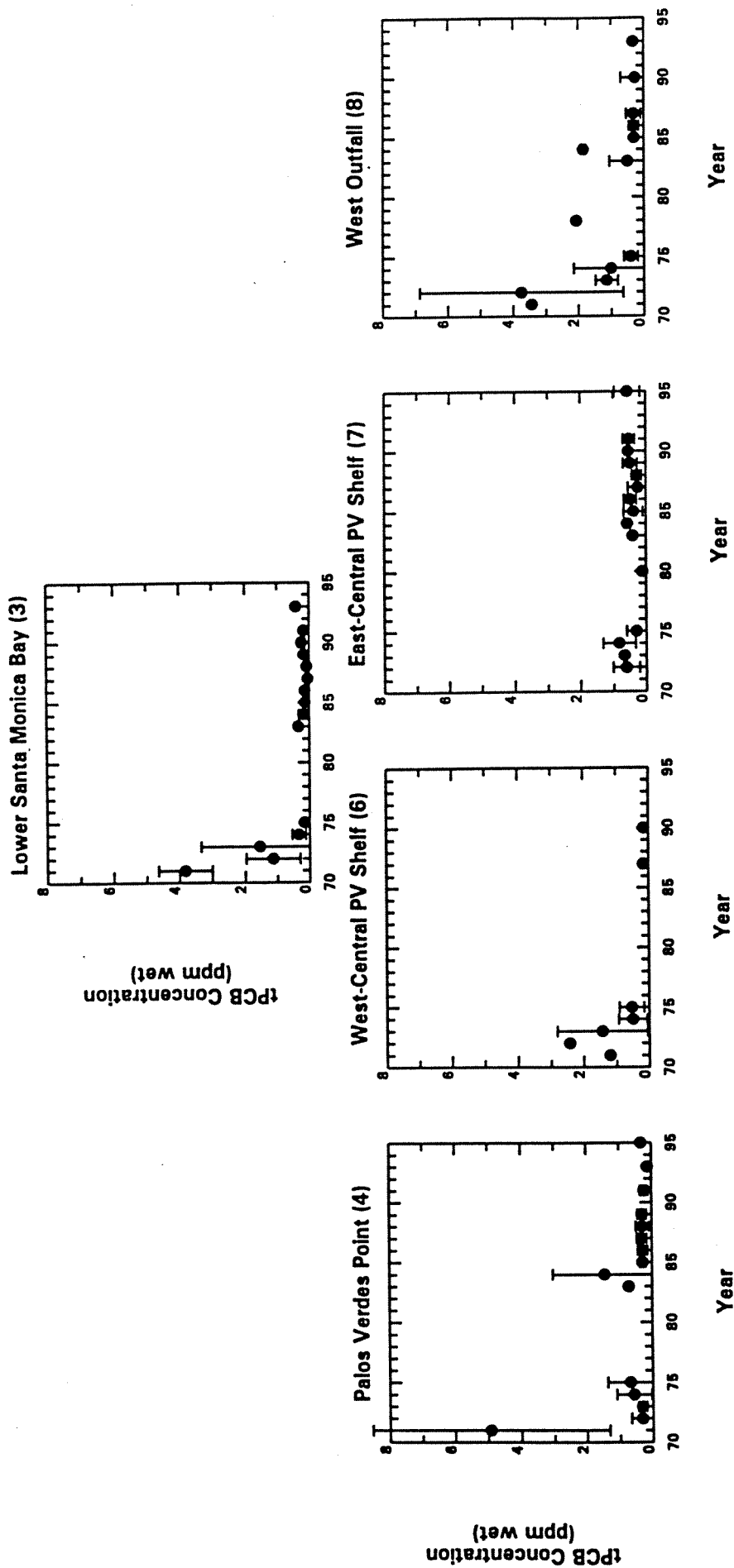
Figure 2-18. Temporal profiles of total PCB concentrations in Dover sole for selected segments of the Southern California Bight (ppm wet weight; +/- 2 standard errors of the mean).
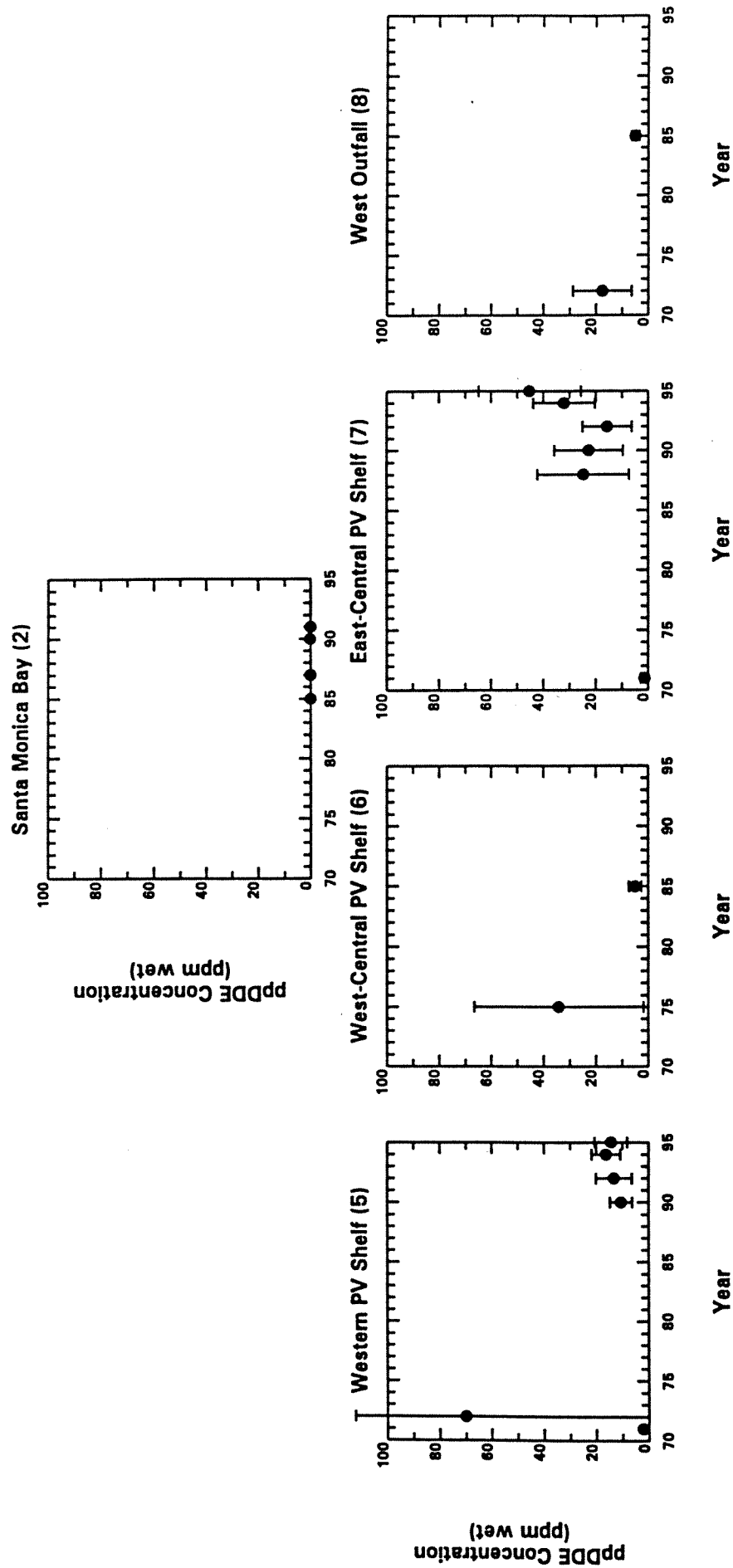
Figure 2-19. Temporal profiles of p,p'DDE concentrations in white croaker for selected segments of the Southern California Bight (ppm wet weight; +/- 2 standard errors of the mean).
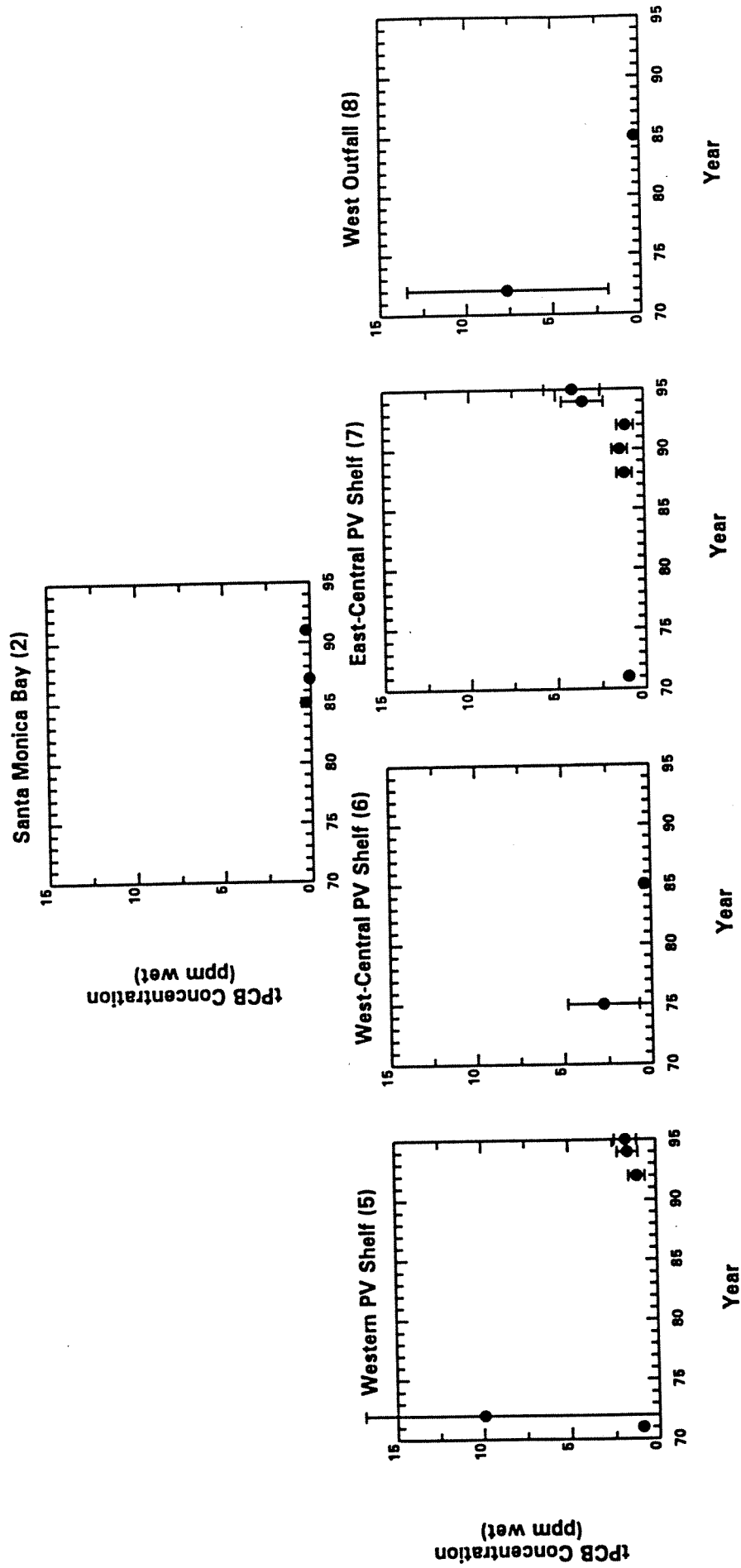
Figure 2-20. Temporal profiles of total PCB concentrations in white croaker for selected segments of the Southern California Bight (ppm wet weight; +/- 2 standard errors of the mean).

Table 2-3. Sea Lion Data in HydroQual Database
Parturient Females from San Miguel Island

Reference	Description	Number of Data Points	Year Sampled
Delong *et al* (1973)	Premature	6	1970
	Full Term	4	1970
Gilmartin (1976)	Premature	10	1972
	Full Term	10	1972
Costa *et al.* (1994)	Premature	10	1991
	Full Term	10	1991
Overall		50	1970-91

The third study was conducted in 1991 as part of the Southern California Bight Damage Assessment (Costa *et al.* 1994). Ten premature parturient and ten full-term parturient females were collected. Blubber samples were analyzed two times for p,p' DDE, PCBs and lipid content by GERG and once by the University of California (UC). The first GERG analysis (Old GERG) and the UC analysis were conducted on splits of homogenized tissue. The second GERG analysis (New GERG) was conducted on the stored extracts from Old GERG. Comparison among the resulting data indicates general agreement, although the Old GERG contaminant results tend to be lower and the UC results exhibit greater variability (particularly for lipid content). For purposes of the model the Old GERG data were used. Because Old GERG contaminant concentrations are about 50 to 70 percent lower than New GERG values (see Table 2-4) the New GERG data were used to establish supposed upper bound estimates of sea lion prey contaminant concentrations.

The 1991 results are in general agreement with the previous studies. The premature parturient females had a mean age of 6.7 years, whereas the full-term parturient females had a mean age of 11.1 years. Mean p,p'DDE and PCB concentrations were about 4 to 5 times higher in the premature parturient females.

The data from the three studies are summarized in Table 2-4. The consistency among the studies is striking. Comparing the early 1970's and 1991, the ratios of premature to full-term parturient female p,p'DDE and PCB residues are similar over concentration drops of factors of about 10 to 20. The age differences between the groups of females are nearly identical. The clear conclusion is that older females that had full-term pups have four to eight times lower p,p'DDE and PCB concentrations than younger females that are not reproducing.

The difference in residues between the two groups has been hypothesized to be due to differences in dietary p,p'DDE and PCB concentrations resulting from utilization of different feeding areas by young and old animals (DeLong *et al.* 1973; Gilmartin *et al.* 1976). This hypothesis requires that the young animals eliminate contaminants rapidly after switching to a low contaminated diet in order to achieve the body burden-age pattern evident in the data. Such a rapid loss can occur in adult females by nursing. Laboratory studies with mammals exposed to PCBs have demonstrated that a large fraction of the body burden of females is lost through milk during one lactation cycle (Gallenberg and Vodicnik 1987; Montesissa *et al.* 1992).

Further insight into the relationship between body burden and reproduction is obtained by examination of the individual measurements (Figures 2-21 and 2-22). It is evident from the figures that the data do fall into high and low concentration groupings with the premature parturient females in the high group and the full-term parturient females in the low group. However, a few of the premature parturient females fall within the low concentration grouping, particularly in the 1991 data set. Because of this, the concentration variability of the premature parturient females is much larger than that of the full-term parturient females. Full-term parturient females are a relatively homogeneous group that some premature parturient females fall within. This pattern may be explained by the impact of nursing. Once a female nurses successfully, her body burden drops rapidly due to loss through the milk. Therefore, all full-term parturient females should have concentrations lower than those in females who have not nursed. If some premature parturient females have had previous successful pregnancies, they would have low contaminant concentrations similar to the full-term group. This could be the case for five of the premature parturient females captured in 1991.

Table 2-4. Summary of p,p'DDE and PCB Residues in the Blubber of Female Sea Lions from San Miguel Island

Year	Premature Parturient Females			Full-Term Parturient Females		
	mean age years	mean p,p'DDE ppm fat	mean PCB ppm fat	mean age years	mean p,p'DDE ppm fat	mean PCB ppm fat
1970	8	944	138	12	109	20
1972	7.7	779	71	11.5	100	16
1991*	6.7	40(60)	14(20)	11.1	10(17)	3(5)

*Numbers outside parentheses are Old GERG. Numbers inside parentheses are New GERG. Note that p,p'DDE comprises approximately 90 Percent of the total DDT in these samples.
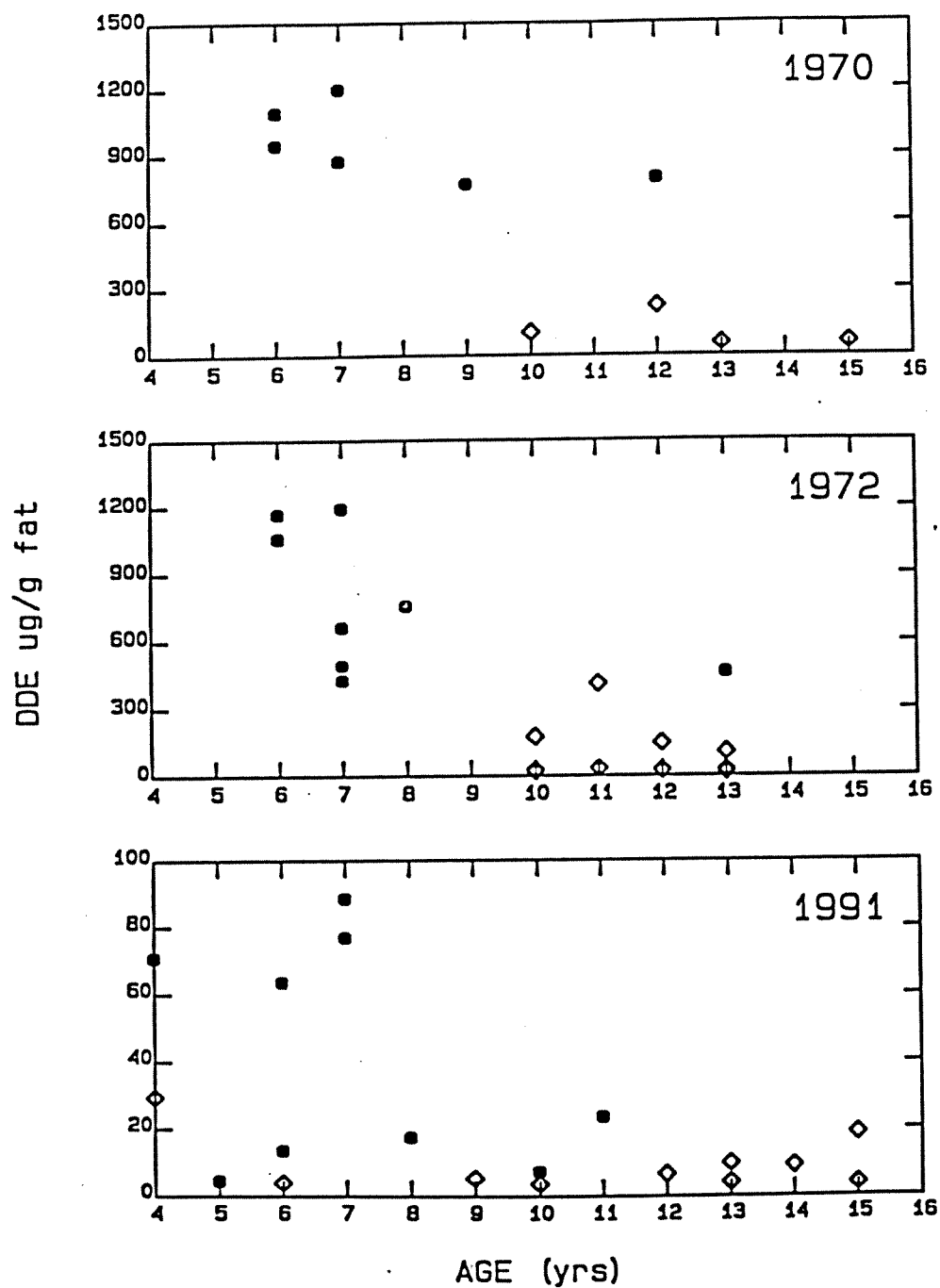
Figure 2-21. Age profiles of DDE concentrations in premature (●) and full-term (◇) parturient female sea lions from San Miguel Island (ppm fat). 1991 values are from Old GERG.
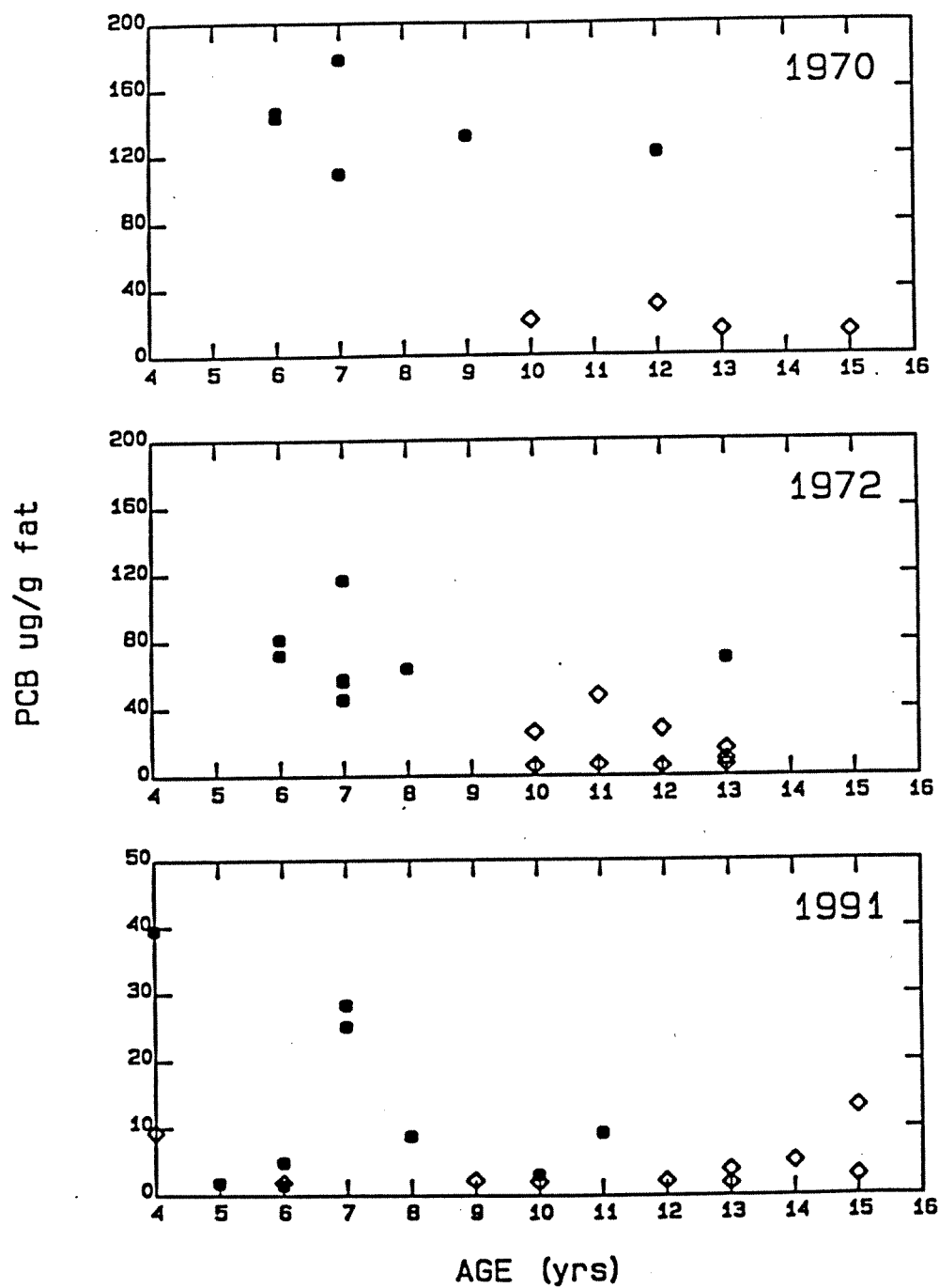
Figure 2-22. Age profiles of Total PCB concentrations in premature (●) and full-term (◇) parturient female sea lions from San Miguel Island (ppm fat). 1991 values are from Old GERG.

2.8 BIRDS

Eggs of peregrine falcons, bald eagles and double-crested cormorants were sampled from the islands in the Southern California Bight. Measured data were adjusted to give equivalent concentrations in freshly laid eggs using egg fresh weights estimated by the collectors of the data (Grainger Hunt, David Garcelon and Michael Fry). Average values for fresh weight-adjusted p,p'DDE and Total PCB concentrations are given in Table 2-5. The values in Table 2-5 were compared with model calculations (see Section 5).

The peregrine falcon eggs were collected from a total of seven clutches, so several eggs came from the same clutches. When the average contaminant levels are recalculated by averaging clutch means, the resulting values are 21.2 +/- 13.6 (n=7) for p,p'DDE and 5.84 +/- 2.99 (n=7) for PCBs. These are not significantly different from the averages in Table 2-5.

Table 2-5. Concentration of p,p'DDE and PCBs in Eggs of the Species of Interest Collected in the Southern California Bight

Species	Location	p,p'DDE[1]	PCBs
Peregrine Falcon	Anacapa, S. Cruz, S. Rosa, S. Miguel	20. $\pm$ 13.(16)	5.8 $\pm$ 3.0(16)
Bald Eagle	S. Catalina	37. $\pm$ 15.(10)	8.1 $\pm$ 3.9(10)
Double-crested Cormorant	Anacapa	8.0 $\pm$ 7.1(13)	3.1 $\pm$ 3.7(13)
	S. Barbara	1.2 $\pm$ 0.42(4)	0.30 $\pm$ 0.07(4)

Notes: All values are corrected to fresh weight.
[1]Values are arithmetic mean $\pm$ standard deviation (number of observations)

Contaminant levels measured in bald eagle eggs from Santa Catalina Island are greater than those measured in peregrine falcon eggs from the northern channel islands by a factor of 1.5 to 2. Both are greater than levels measured in cormorant eggs.

The distributions of contaminant levels in each species are presented in Figures 2-23, 2-24 and 2-25[5]. Some of the data are normally distributed, while some appear to be better described by a lognormal distribution. One high outlier was measured in a peregrine falcon egg collected on Anacapa Island. Contaminant levels in double-crested cormorant eggs from Anacapa Island are greater than levels measured at Santa Barbara Island by about a factor of 7 (p,p'DDE) and 10 (total PCBs).

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[5]Probability plots are described, for example, by Wine (1964).
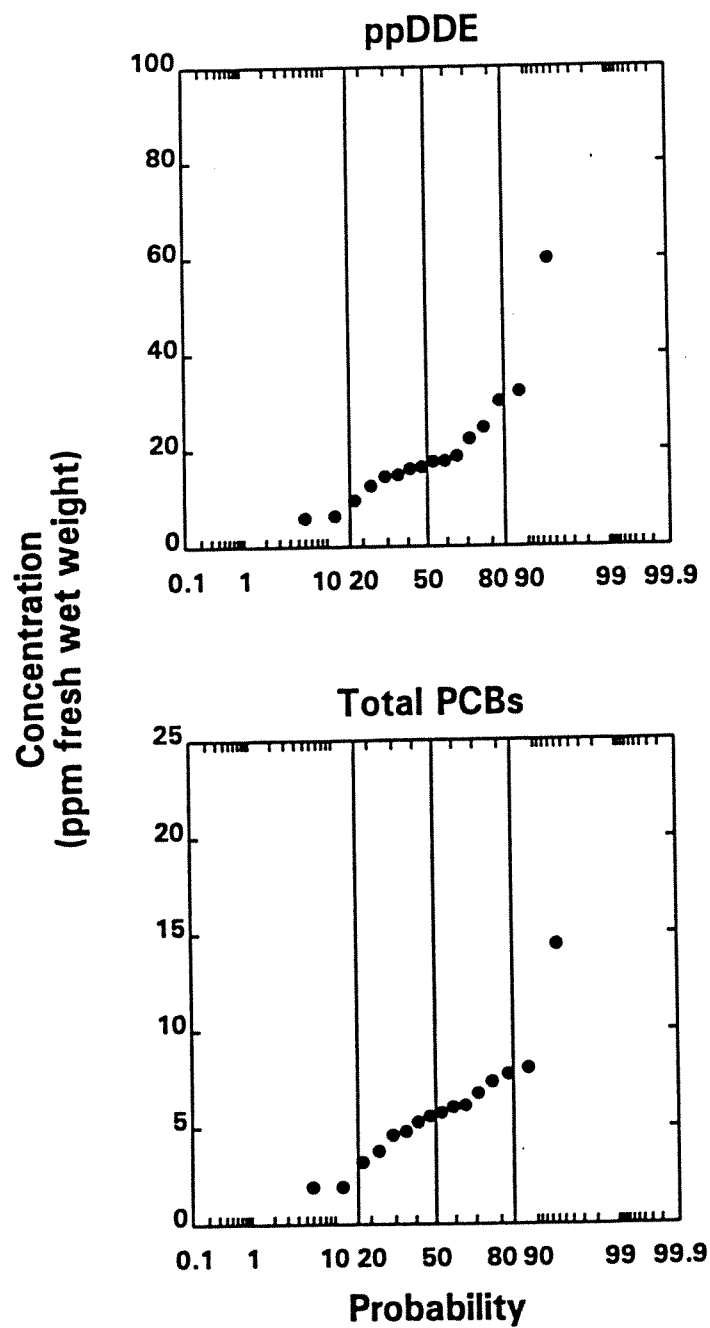
ppDDE

Total PCBs

Anacapa, S. Cruz, S. Rosa and S. Miguel Islands

Figure 2-23. p,p'DDE and total PCB concentrations in eggs of peregrine falcons from the Southern California Bight (ppm fresh wet weight).

ppDDE

Total PCBs

Probability

Concentration
(ppm fresh wet weight)
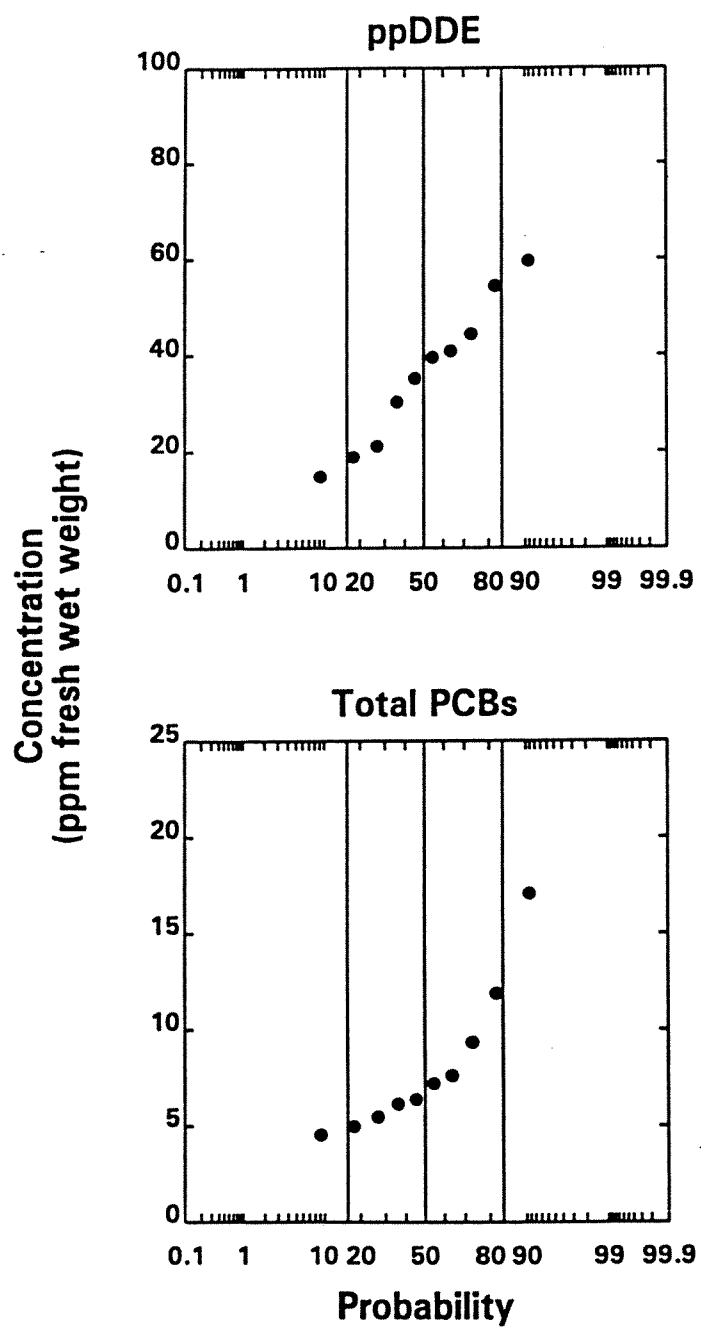
Santa Catalina Island

Figure 2-24. p,p'DDE and total PCB concentrations in eggs of bald eagles from the Southern California Bight (ppm fresh wet weight).

ppDDE

Anacapa Island

Santa Barbara Island
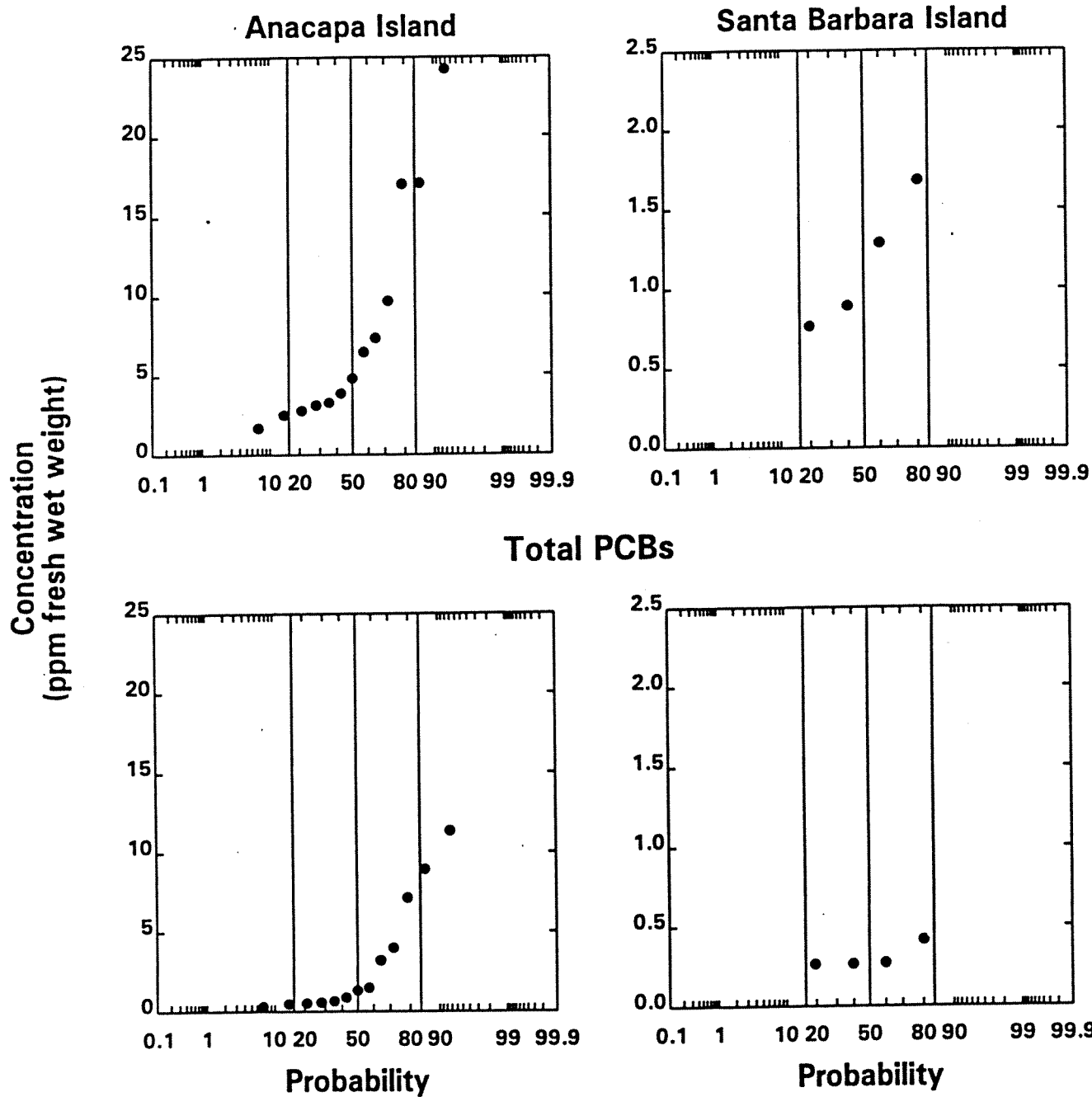
Total PCBs

Figure 2-25. p,p'DDE and total PCB concentrations in eggs of Double-crested cormorants from the Southern California Bight (ppm fresh wet weight).

SECTION 3

FOOD CHAIN MODELS OF FISH

3.1 INTRODUCTION

Analyses of p,p'DDE and PCB data indicate that concentrations in sediments, water and fish are elevated in the region off the Palos Verdes Shelf. To assess whether food web transfer of the contaminants present in local sediments and water in this region could account for levels measured in local fish, bioaccumulation models of these two contaminants were developed for white croaker, Dover sole and kelp bass. First, the models were applied to each fish species in one HydroQual segment near the Whites Point outfall. Next, the models were applied to each fish species in a region extending from 11 km south of the outfall to 14 km north of the outfall, a region over which there is a gradient in contaminant levels. The goal of the modeling effort was to assess whether exposure to local, relatively highly contaminated water and sediment is necessary to explain the levels observed in the fish, or whether exposure to levels characteristic of areas beyond the Palos Verdes Shelf is sufficient. The concentrations of p,p'DDE and PCBs in the sediment and water to which the fish are exposed were derived from the data described in Section 2. Based on the density of observations, sediment concentrations from segment 7 and mussel concentrations from segments 8 and 9 were used for the first set of simulations. Mussel concentrations exhibit a weaker gradient than sediments (see Section 2), so exact matching of locations was not necessary. Annual averages for these segments were used to establish continuous temporal concentration profiles for p,p'DDE (1975 to 1995) and for PCBs (1980 to 1995). The estimation of exposure levels for the simulations in the region extending from 11 km south of the outfall to 14 km north of the outfall is described in section 3.5.3.

In addition, several simulations were performed to explore the sensitivity of the results to variation in specific parameters: Dover sole growth rate, kelp bass migration, and mussel bioaccumulation factors. Finally, the overall uncertainty in model results due to uncertainty in model parameter values was assessed in a Monte Carlo analysis.

3.2 DEVELOPMENT OF EXPOSURE LEVELS FOR THE ZONE OF HIGH CONTAMINATION

3.2.1 Sediments

Annual average sediment p,p'DDE concentrations in segment 7, are shown in Figure 3-1, along with a visually defined continuous profile. Concentrations are represented as declining exponentially from about 3250 ppm organic carbon (ppm OC) in 1970 to about 260 ppm OC in 1982. From 1982 to 1995 concentrations remain constant at 260 ppm OC.

PCB levels in segment 7 in the 1980's averaged 36 ppm OC and showed no temporal trends. There were no data available in the 1970's, so no attempt was made to estimate concentrations before 1980. The exposure level used in the models is shown in Figure 3-2.

3.2.2 Water

Water column p,p'DDE and PCB concentrations were estimated from the concentrations in intertidal mussels and an estimated bioaccumulation factor (BAF). BAF values were determined as described in Section 2.5 (8.9×10^6 for p,p'DDE and 2.0×10^6 for PCBs). The mussel data were treated the same as the sediment data: annual averages were used to establish a continuous temporal profile. Data from segments 8 and 9 were used to define water column concentrations. For p,p'DDE an exponential decline from 3 ppm wet weight (55 ng/l in the water) in 1970 to 0.12 ppm wet weight (2 ng/l) in 1982 is followed by an exponential decline to 0.1 ppm wet (1.8 ng/l) in 1995 (Figure 3-3). For PCBs, an exponential decline from 0.026 ppm wet (29 ng/l) in 1981 to 0.017 ppm wet (1.6 ng/l) in 1995 is used (Figure 3-4).

Because the mussels are exposed to nearshore water column concentrations, we have assumed that these concentrations are representative of concentrations at the offshore sites where the fish were captured. This assumption is justified by measurements of concentrations in mussels suspended in the water column near the outfall for 24 weeks (total DDT and Total PCBs; Young 1982) and for 3 months (p,p'DDE and Total PCBs; Martin et al. 1984). The contaminant concentrations in these mussels are similar to intertidal mussels collected in the same year. The Young study was conducted in 1974 and the results are shown as the square in Figure 3-3. The Martin et al. study was conducted in 1983 and the results are shown by the inverted triangles. Young suspended mussels at 5 depths and found that concentration increased
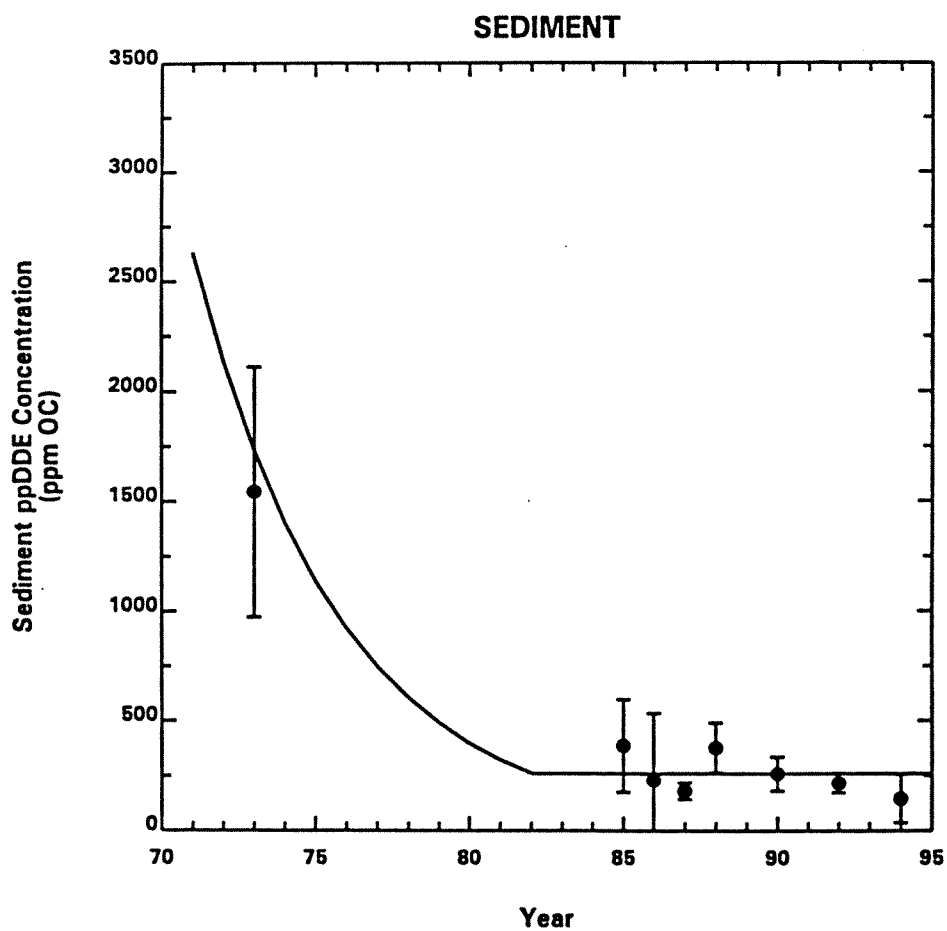
SEDIMENT

Figure 3-1. Temporal profile of p,p'DDE concentrations in surface sediments from zone of high contamination (segment 7; ppm organic carbon; arithmetic mean +/-2 standard errors of the mean). Symbols indicate data; lines indicate exposure concentrations used in model.
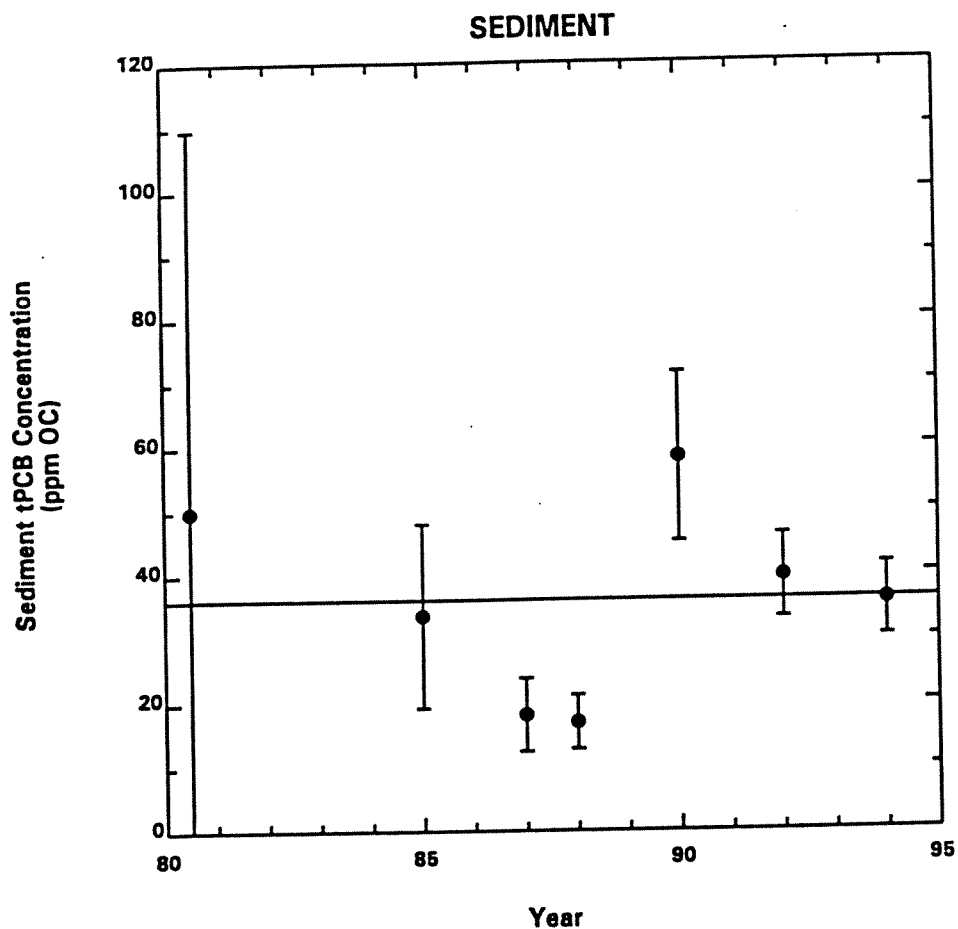
SEDIMENT

Figure 3-2. Temporal profile of Total PCB concentrations in surface sediments from zone of high contamination (segment 7; ppm organic carbon; arithmetic mean +/- 2 standard errors of the mean). Symbols indicate data; lines indicate exposure concentrations used in model.
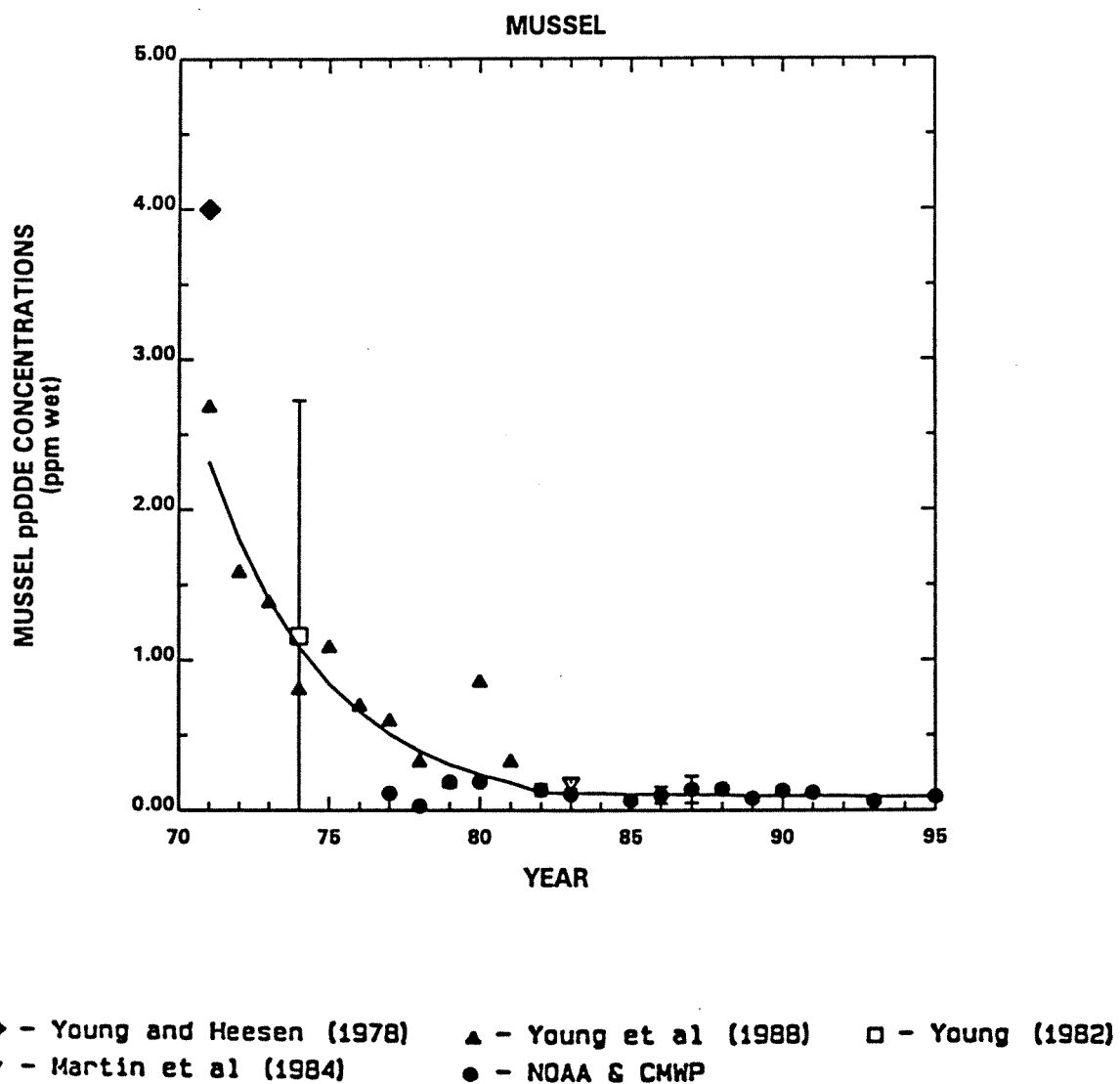
Figure 3-3. Temporal profile of p,p'DDE concentrations in mussels for the zone of high contamination (segments 8 and 9; ppm wet weight; arithmetic mean +/- 2 standard errors of the mean). Symbols indicate data; lines indicate exposure concentrations used in model.

MUSSEL

∇ – Martin et al (1984)   ● – NOAA & CMWP
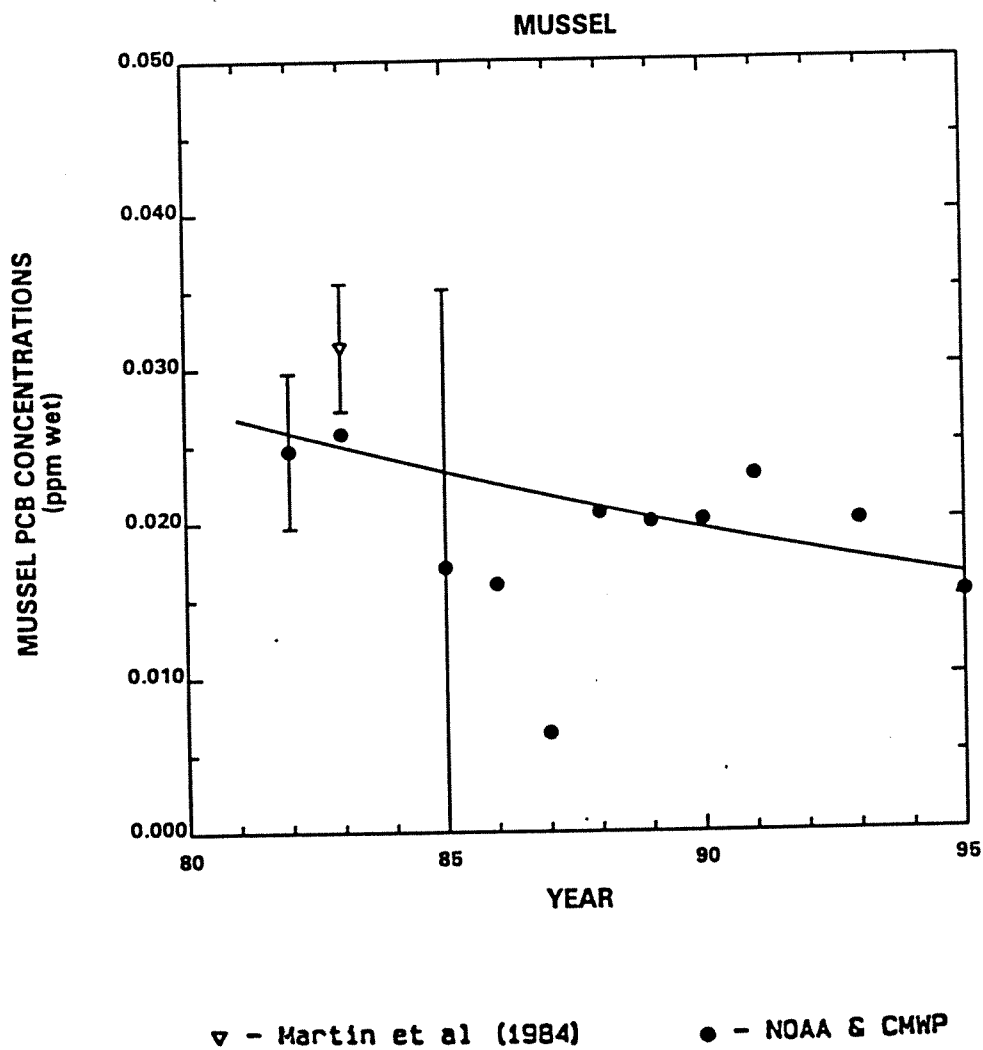
Figure 3-4.   Temporal profile of Total PCB concentrations in mussels for the zone of high contamination (segments 8 and 9; ppm wet weight; arithmetic mean +/- 2 standard errors of the mean). Symbols indicate data; lines indicate exposure concentrations used in model.

with depth (Figure 3-5); the range was about a factor of 5.

3.3   ALGAE AND INVERTEBRATES

Macroalgae, phytoplankton, benthic invertebrates, zooplankton and kelp bed/hard bottom invertebrates are components of the food webs of the fish. The transfer of contaminants to these organisms was modeled as follows.

For the macroalgae and phytoplankton, a simple partitioning with dissolved contaminant in the water was assumed. The partition coefficients of a variety of hydrophobic chemicals between water and the aquatic macrophyte Myriophyllum was studied by Gobas *et al.* (1991). They found that the lipid-based plant-water partition coefficient was approximately equal to K_{ow} for chemicals with a wide range of K_{ow}'s; Myriophyllum is a vascular plant. Nonvascular plants such as kelp are expected to exhibit equal or more rapid equilibration with water, because vascular plants contain additional structural tissues which may act to hinder equilibration. Therefore, the equilibrium partitioning method was applied here. Using the average kelp lipid content of 0.003 g lipid/g wet weight (Young *et al.* 1988b), and K_{ow} values of 6.96 for p,p'DDE and 6.3 for PCB's, the partition coefficients for p,p'DDE and PCB's between water and macroalgae used in the model are 27,000 and 6,000 L/Kg wet.

Phytoplankton was modeled similarly to macroalgae, with a partition coefficient of 40,000 L/Kg wet weight, based on data reviewed by Connolly (1991).

Data on bioaccumulation of PCBs by benthic macroinvertebrates were collected from published literature. Values of the Biota/Sediment Accumulation Factor (BSAF) measured in oligochaetes are given in Table 3-1. Values for p,p'DDE and for Total PCBs do not appear to differ (Table 3-1). An overall average bioaccumulation factor of 2.1 g OC/g lipid was calculated and used for the BSAF in the white croaker and Dover sole model.

SUSPENDED MUSSELS
NEAR PALOS VERDES OUTFALL
JUNE–NOVEMBER, 1974

TOTAL DDT

WATER COLUMN DEPTH (M)

PERCENT OF TOTAL DEPTH

TOTAL PCB

WATER COLUMN DEPTH (M)

PERCENT OF TOTAL DEPTH

EQUILIBRIUM CONCENTRATION
(MG/KG DRY)
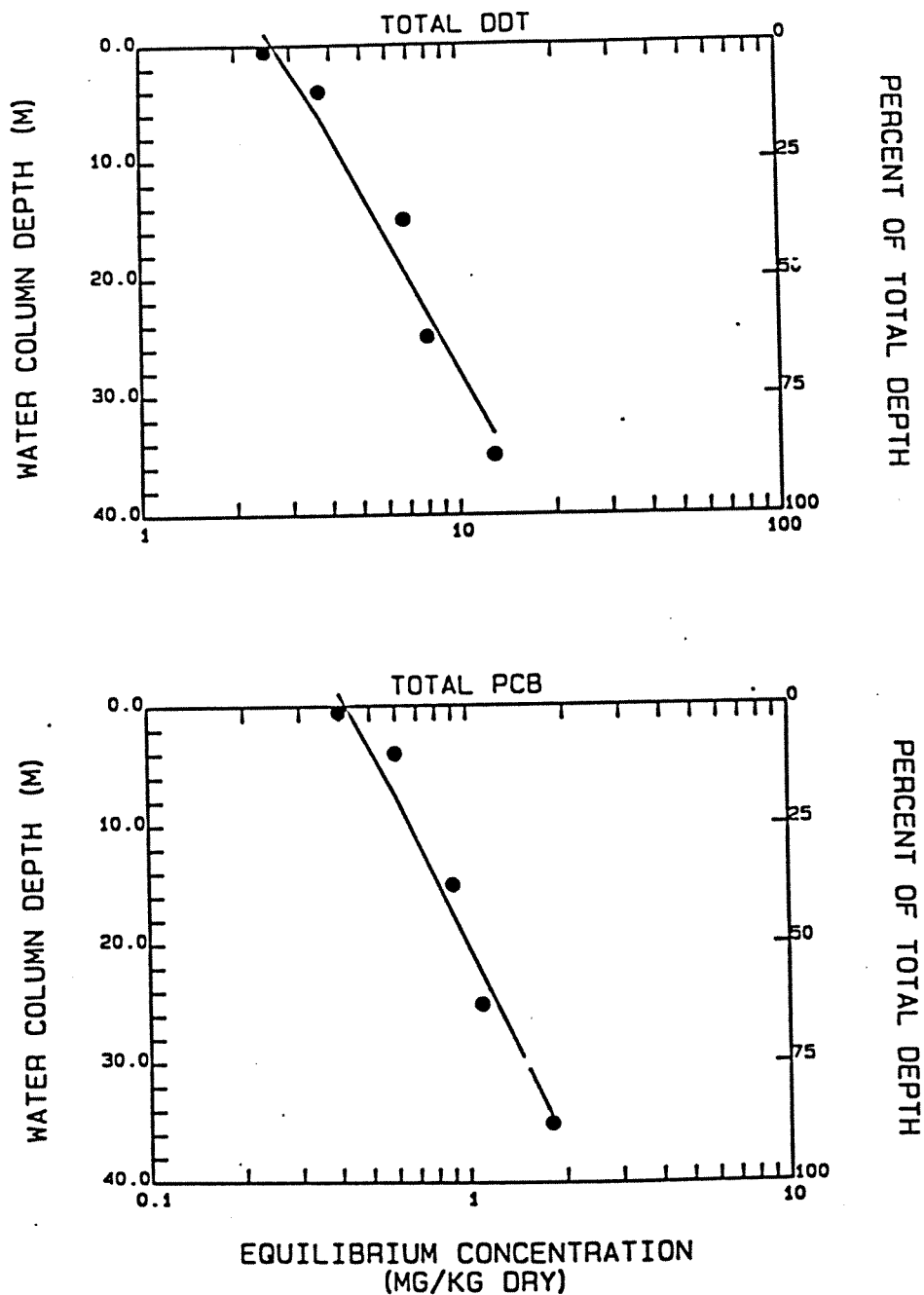
Young. 1982. in "Ecological Stress and the New York Bight. pp.263-276

Figure 3-5. Total PCB and total DDT levels in suspended mussels from the Palos Verdes region in relation to water column depth.

Table 3-1. Biota/Sediment Accumulation Factors for Macroinvertebrates

Chemical	Group	Location	BSAF (g OC/g lipid) Mean	+/- std dev	Reference
p,p'DDE	oligochaete	field	1.5	+/- 0.43	Oliver 87
p,p'DDE	oligochaete	lab	4.6		Oliver 87
Total PCBs	oligochaete	field	1.7	+/- 0.8	Oliver 87
Total PCBs	oligochaete	lab	4.3	+/- 1.7	Oliver 87
Total PCBs	oligochaete	field	0.87	+/- 0.38	Ankley et al. 92
Total PCBs	oligochaete	lab	0.84	+/- 0.35	Ankley et al. 92
Total PCBs	oligochaete	field	0.84		Oliver & Niimi 88
overall average			2.1		

Zooplankton and kelp bed/hard bottom invertebrates were explicitly modeled, with body burdens considered to be in steady-state with respect to dissolved water concentrations. Values for fraction lipid and fraction dry weight for zooplankton were based on values for copepods measured by Vanderploeg et al. (1992). A lipid fraction of 0.015 (g lip/g wet total weight) was assumed for kelp bed/hard bottom invertebrates, based on data collected for lobsters in New Bedford Harbor, Massachusetts (Connolly 1991). Net growth efficiency for zooplankton and for kelp bed/hard bottom invertebrates was set at 0.25, the average value reported by Humphreys (1979) for non-insect invertebrates.

Tissue composition values for algae and invertebrates are given in Table 3-2.

Table 3-2. Tissue Composition for Algae and Invertebrates

	Fraction Dry	Fraction Lipid	Source
Kelp	0.08	0.003	Young et al. 1988b
Kelp bed/hard bottom invertebrates	0.19	0.015	Connolly 1991
Polychaetes	0.15	0.015	Connolly 1991
Zooplankton	0.20	0.040	Vanderploeg et al. 1992

3.4. ESTIMATION OF TOXICOKINETIC PARAMETERS

3.4.1 Gut Transfer

The fraction of ingested contaminant that is transferred across the gut wall and into the animal is termed the assimilation efficiency. A compilation of experimental estimates of assimilation efficiency of various PCB congeners has shown that values ranges from about 0.1

to 1 (Connolly *et al.* 1992, Parkerton 1993). To examine these data, multiple values from individual studies were averaged to give all studies equal weight; the data were then grouped into half log unit K_{ow} bins (eg., 4.25 to 4.75) and displayed as box plots (Figure 3-6). Congeners with log K_{ow} values below 6.75 (generally mono- to pentachlorobiphenyl) have similar assimilation efficiencies, although a slight decline with increasing K_{ow} is evident. Median values range between 0.75 and 0.85. As log K_{ow} increases beyond 6.75 a more dramatic decline occurs. Values for DDT (including a single number for DDE) range from 0.43 to 0.85 (Table 3-3) with a mean of 0.60 and a standard error of 0.066. These values are consistent with values for PCBs of similar log K_{ow} (~7).

Table 3-3. Dietary Assimilation Efficiency of p,p'DDT[1] and p,p'DDE[2] in Fish

Species	Diet Type	Mean Assimilation Efficiency	Reference
Rainbow trout	synthetic	0.64[1]	Muir and Yarechewski 1988
Rainbow trout	synthetic	0.67[2]	Niimi and Oliver 1988
Saltmarsh killifish	fish	0.43[1]	Vetter 1983
Channel catfish	invertebrate	0.85[1]	Ellgehausen *et al.* 1980
Brook trout	synthetic	0.43[1]	Macek and Korn 1970
Cod	synthetic	0.55[1]	Mitchell *et al.* 1970

The differences between studies reflect various factors including measurement error, residence time in the gut, digestibility of the portion of the ingested prey that contains most of the contaminant (for hydrophobic organic chemicals this is fat tissue) and the physical-chemical mechanisms responsible for moving the chemical across the gut wall. Of the biological and chemical factors, the digestibility of the prey appears to have the greatest impact. Recent studies indicate that the assimilation efficiency of hydrophobic contaminants is closely linked to dietary assimilation of lipids (Van Veld 1990). Our previous modeling studies (see citations in Table 1-1) have suggested that assimilation efficiency of hydrophobic organics with log K_{ow} values below about 6 to 6.5 is similar to that of food energy and is perhaps equal. This suggestion is supported by experimental studies with metals and radionuclides which indicate a direct correspondence between the uptake of contaminant by zooplankton and the fraction in the digestible component of the algal diet (Reinfelder and Fisher 1991). At the K_{ow} of p,p'DDE (10^7) the measurements cited above and our study of New Bedford Harbor PCBs indicate that assimilation efficiency is approximately 20 to 40 percent lower than the food energy assimilation efficiency (assuming that food energy assimilation efficiency is about 0.8). Based on the above information, we have maintained equality between contaminant and food assimilation efficiency
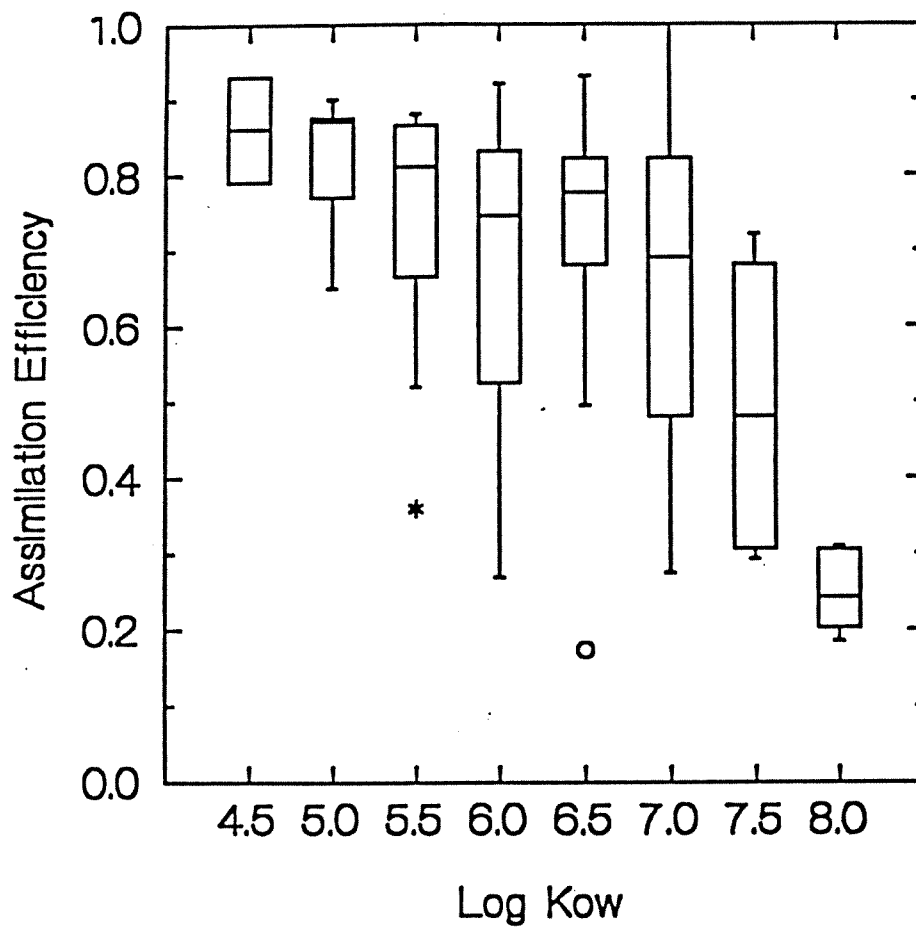
Figure 3-6. Dietary assimilation efficiencies of PCBs in fish (box plots).

for total PCBs and used a contaminant: food energy assimilation efficiency ratio of 0.75 for p,p'DDE. Note that the model requires only the ratio of these coefficients and not their absolute values.

3.4.2 Gill Transfer

The transfer for contaminants between blood and water across the gill epithelia is determined from the transfer of oxygen and a ratio between the uptake efficiencies of contaminant and oxygen. A compilation and analysis of published laboratory measurements (Connolly *et al.* 1992) indicates that over a broad range of hydrophobicity (log K_{ow} from 3 to 7) this ratio is of order 1 (Figure 3-7). The ratio tends to decline above and below this log K_{ow} range. This data compilation includes several studies of PCBs and one study of DDT. The average ratios are 0.99 for PCBs and 1.18 for DDT. A value of 1 is used for all species in the model.

3.4.3 Octanol/Water Partition Coefficient

The octanol/water partition coefficient (K_{ow}) is a key parameter used to define the partitioning of contaminant between lipid and aqueous phases. This affects the rates of excretion of contaminants. There have been several experimental estimates of K_{ow} for p,p'DDE. In determining sediment quality criteria, the U.S.E.P.A. chose the slow-stir flask method for determining K_{ow} values because of its relatively low variability and low bias relative to other methods (E.P.A. 1993). The value of De Bruijn *et al.* (1989) was estimated using this technique; therefore, their value (Log K_{ow} = 6.96 L/Kg octanol) is used for p,p'DDE. This value is one of the most recent estimates in the literature and, in addition, agrees with the results of the calculation performed using the Medchem CLOGP program version 3.53 (6.94; see De Bruijn *et al.* 1989).

The appropriate Log K_{ow} for PCBs is dependent on the congener composition in the animals. Values estimated directly from the congener-based data for white croaker (6.26 for segment 5 and 6.19 for segment 9; Santa Monica Bay Restoration Project) and for kelp bass (6.36 for segment 3, Costa *et al.* 1994) were used to establish an average value of 6.3. This value is equivalent to the Log K_{ow} for Aroclor 1254.
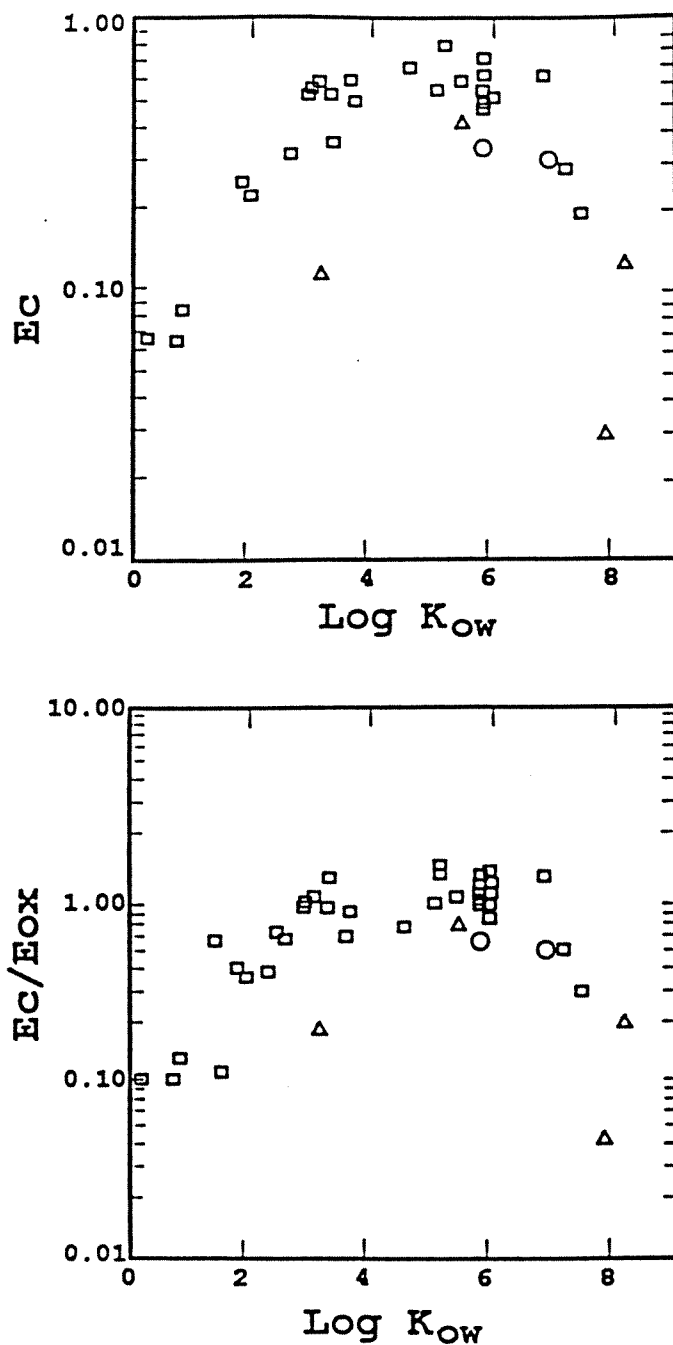
(a) Transfer Efficiency of Non-Ionic Organic Chemicals Across the Fish Gill as a Function of Log K_{ow}.

(b) Relative Transfer Efficiency of Contaminant to that of Oxygen Across the Fish Gill as a Function of Log K_{ow}. Symbols denote fish weight classes: circles< 10g; triangles 10-100g; squares >100g.

Figure 3-7. Transfer efficiency of non-ionic organic chemicals across the fish gill as a function of log K_{ow}.

3.5   WHITE CROAKER MODEL

3.5.1 Development of Species-Specific Physiological Parameters

3.5.1.1   Habitat

White croaker eggs and larvae are pelagic, juveniles are limited to shallow nearshore waters, and adults move to deeper water (Love *et al.* 1984). Both the juveniles and adults are benthic, preferring a sandy, featureless bottom environment ranging from the surf zone to maximum depths of 183 meters. Tolerant of degraded environments, white croaker are often abundant around sewage outfalls (Love *et al.* 1984). They are considered an important sportfish caught mainly from piers and small boats in southern California (Ware 1979 and Love *et al.* 1984).

3.5.1.2   Food

The larvae feed on zooplankton (Ware 1979). The young and adults are benthic feeders foraging primarily at night in tight columnar schools over open sand bottoms. White croaker are omnivorous feeders who locate their prey by touch. When prey is located, the fish sucks up the prey and sediments from the bottom ingesting both without chewing (Allen 1982 and Ware 1979). The adult diet consists mainly of polychaetes and crustaceans (Ware 1979 and Malins *et al.* 1987). In the model, young of the year white croaker feed on zooplankton. After age one, white croaker feed on benthic invertebrates (Figure 3-8).

3.5.1.3   Movement and Migration

The eggs and larvae are pelagic and distributed in a narrow band along the coast (22 to 36 meter depth) (Love *et al.* 1984). The juveniles are limited to shallow nearshore waters (3 to 6 meter depth) and move to deeper water as adults (Love *et al.* 1984).

p,p'DDE levels in white croaker exhibit a sharp decline away from the outfall suggesting that movement along the shelf is limited (Figure 2-13). If movement was extensive, a more homogeneous distribution of contaminant levels in the white croaker populations would be expected. In the model, white croaker are assumed to remain in the zone of high contamination (segment 7) throughout their lives.
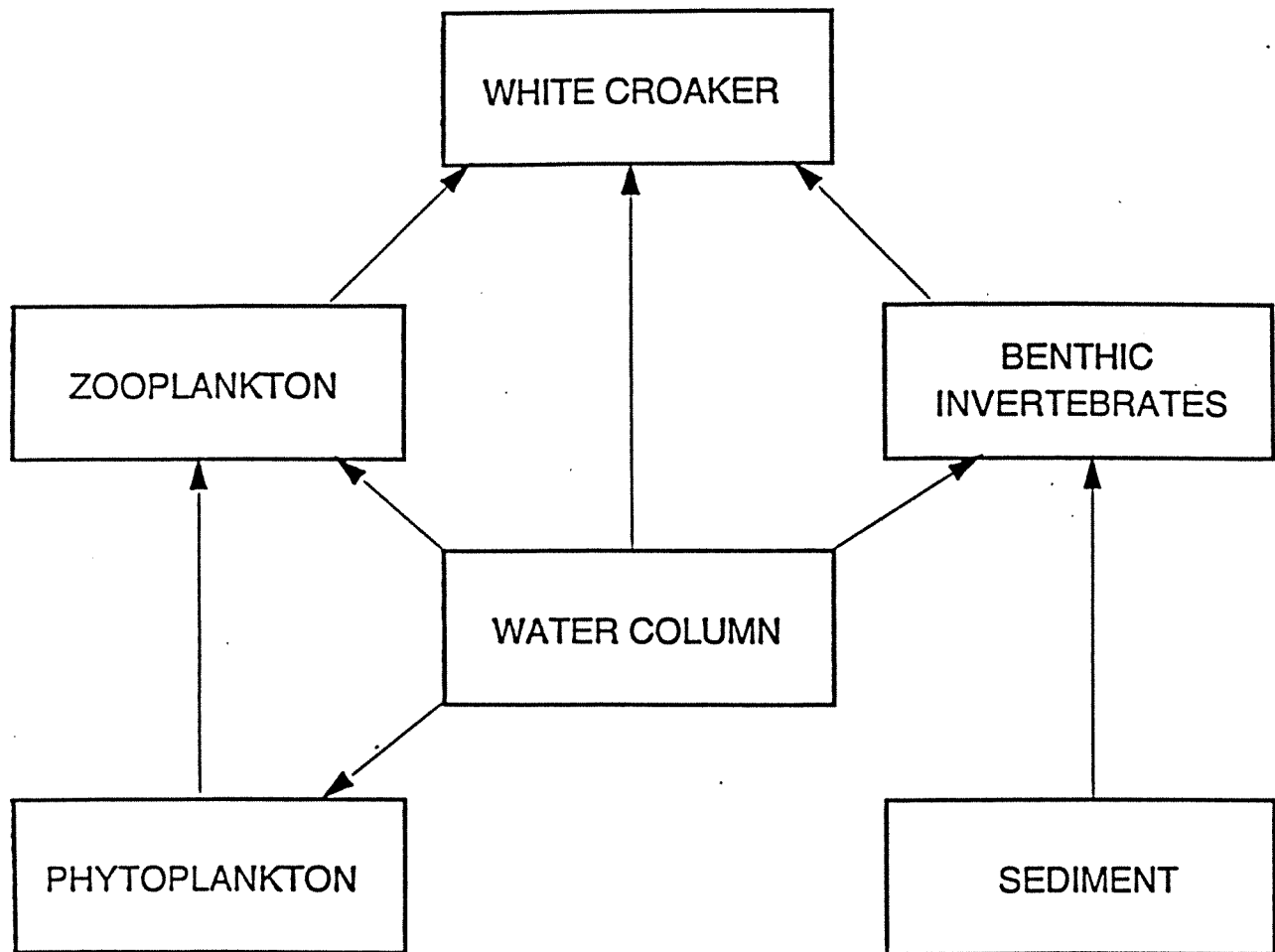
Figure 3-8. White croaker food web.

In addition, the PCB congener data collected as part of the Santa Monica Bay Restoration Project (SMBRP) were used to evaluate spatial variation in contamination characteristics. The value of Log K_{ow} was estimated for Total PCBs in white croaker in several segments. Estimates of the average K_{ow} were made by weighing the congener K_{ow} values of Hawker and Connell (1988; adjusted to equivalence with De Bruijn et al. (1989); see Section 2.6) with the measured congener proportions of SMBRP. Values increased consistently from 6.2 in segment 9 (outfall area) to 6.5 in segment 1 (Santa Monica Bay). The observed increase in Log K_{ow} with distance from the outfall is consistent with our understanding of transport processes for hydrophobic chemicals in aquatic environments; the more hydrophobic congeners are expected to be selectively enriched away from the source of contamination, because the primary mechanism of movement is sediment transport, and because the more hydrophobic compounds partition more strongly to the organic matter on sediment particles. Lower congeners are more readily lost to the water column. Thus, the characteristics of the white croaker contamination change with location, suggesting a lack of movement along the shelf.

3.5.1.4   Growth and Body Composition

Growth rates for white croaker were determined for each age class based on a survey performed in the vicinity of the study area, with age determined by otolith annuli (Love et al. 1984) (Figure 3-9). Otolith analysis provide an estimate of the relationship between fish length and age. This must be converted to a weight/age relationship for use in the model. The length/age relationship measured by Love et al. (1984; upper left panel of Figure 3-9) was combined with the weight/length relationship measured by Isaacson (1964; middle left panel of Figure 3-9) to produce the weight/age relationship presented in the bottom left panel of Figure 3-9. There is good agreement between both Isaacson's and Love's weight-length relationships.

Percent lipid varied greatly in segment 7 (0.5 to 18.0 percent; Figure 3-9, middle right panel). We are unable to determine if this represents variation in the population or is a consequence of the extraction methodology. The appropriate lipid fraction value for the purposes of the model is one that best accounts for the difference between wet weight-based and lipid-based contaminant concentrations. Analysis of the contaminant data suggests an average lipid content of 6 percent (fillet), which is consistent with data on Figure 3-9. The average measured dry weight is 20 percent, with little variation. Muscle dry weight was assumed to represent the whole-body values.

GROWTH INFORMATION

WHITE CROAKER
IN THE SCB

Length/Age Relationship

Length Distribution

Weight/Length Relationship

Percent Lipid/Length Relationship

Weight/Age Relationship

Percent Dry Weight/Length Relationship
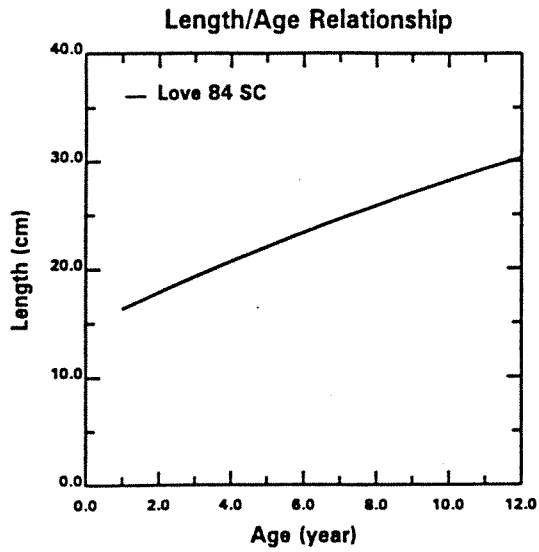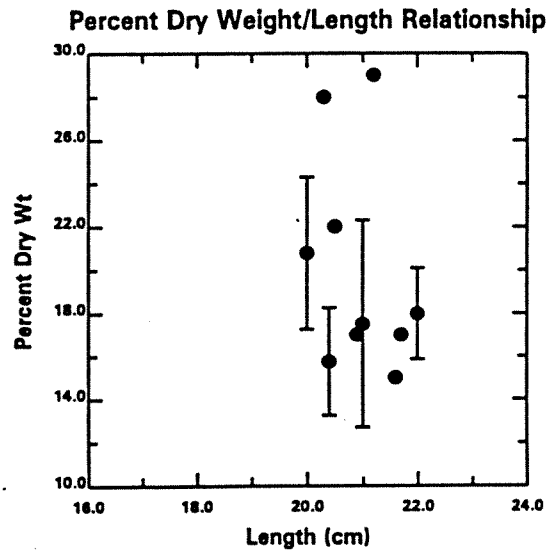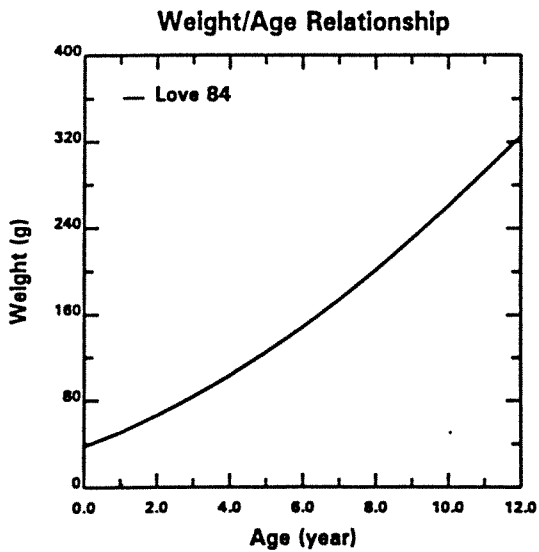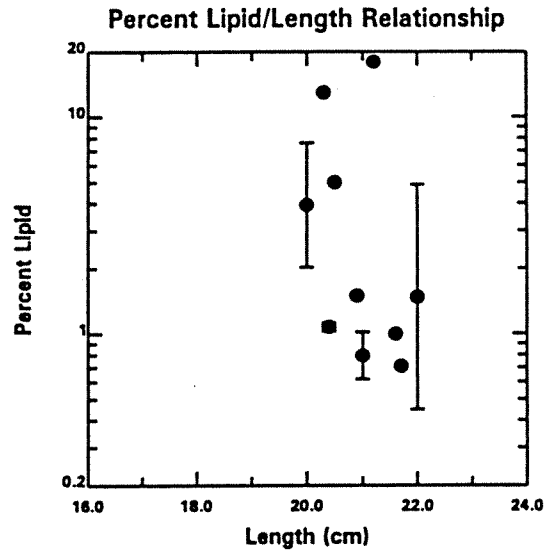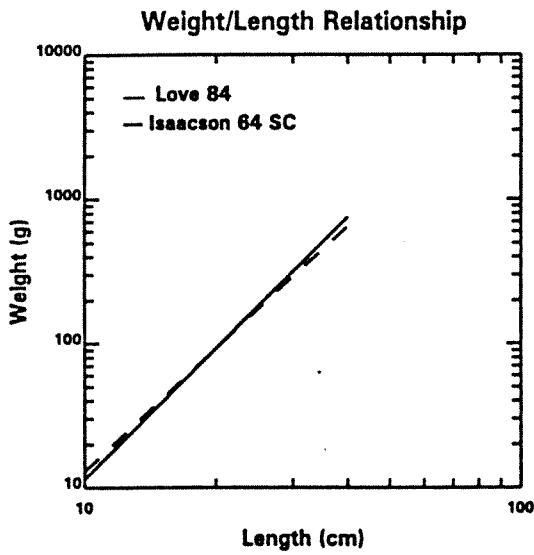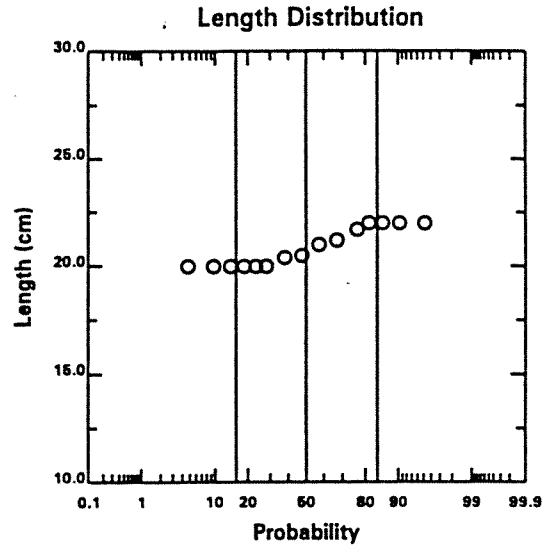
Figure 3-9.   White croaker growth and body composition data

The model calculates contaminant levels on a whole-body basis, and the contaminant data for fish in the HydroQual database are for fillets. Fillet lipid contents for kelp bass, white croaker and Dover sole were calculated using data in the HydroQual database. These lipid contents were then adjusted to arrive at realistic lipid contents for whole fish. Ratios of whole-body/muscle lipid contents have been measured in several species (Table 3-4). One value for winter flounder was 5.5; this value is used here for Dover sole. The other values ranged from 2.0 to 3.0,with one outlier (4.6); the average is 2.6 (n=7). In the present analysis, fillet lipid values were multiplied by 2.6 to arrive at whole-body lipid contents for use in the model. Whole-body concentrations calculated by the model were adjusted by the same ratio to produce equivalent values for fillet in order to compare with the data.

Table 3-4. Whole-Body/Fillet Lipid Ratios

Species	Water Body	Ratio	Reference
coho salmon	Credit River	2.0	1
coho salmon	Lake Ontario	2.5	1
lake trout	Lake Ontario	2.4	1
brown trout	Lake Ontario	3.0	1
rainbow trout	Credit River	2.7	1
rainbow trout	Lake Ontario	4.6	1
largemouth bass	Hudson River	2.5	2
northern pike	Lac Ste. Anne, Alberta	3.0	3
winter flounder	New Bedford Harbor	5.5	4

References:
1 Niimi and Oliver 1989
2 unpublished data
3 Medford and Mackay 1978
4 Connolly 1991

Model results are presented for the age classes of white croaker represented in the database.

3.5.1.5   Respiration

A general relationship between standard respiration and fish weight (Hemmingsen 1960) was used to empirically describe respiration rate as a function of weight and temperature, because species-specific respiration measurements were not available for white croaker. The resting metabolic rate of Hemmingsen (1960) was multiplied by 2 to give routine active metabolic rate (Peters 1983). Net growth efficiency (growth rate divided by the sum of growth

and respiration rates) was used as a check on the reasonableness of the respiration and growth rates. Humphreys (1979) reviewed production and respiration rates in a wide variety of animals and found an average net growth efficiency of 0.10 for the group including fish, with a standard error of 0.009. This is similar to the white croaker values calculated by the model (range 0.05 to 0.08 (Figure 3-10). Figure 3-10 details the relationships between growth, respiration and net growth efficiency as a function of white croaker age.

3.5.2 Model Results

Model simulations were performed for p,p'DDE and total PCBs in a zone of high contamination (i.e. near the outfall). For p,p'DDE, the model was run for a period of 20 years, starting in 1975 and ending in 1995. Due to the paucity of PCB data in the 1970's, model simulations for total PCBs were conducted for a period of 15 years (1980 to 1995). Results of both simulations are presented on wet-weight and lipid-normalized bases.

Predicted lipid-based levels of p,p'DDE and total PCBs in white croaker are similar to values observed in the data, and are within two standard errors of the means for most of the data (Figures 3-11 through 3-14). The relationship between model and data is presented in Appendix C using box plots. This agreement between calculated and observed concentrations are consistent with the hypothesis that p,p'DDE and total PCB levels in white croaker originate from the highly contaminated sediments of the Palos Verdes Shelf.

The relationship between model results and data exhibits more variability on a wet-weight basis than on a lipid basis. The lipid-based contaminant levels provide a better basis for comparing model results with data, because wet weight-based contaminant concentrations are expected to vary with the organism's lipid content. The lipid contents of the 1994 and 1995 fish was greater than that of the pre-1993 fish, resulting in lower wet weight-based concentrations measured before 1993.
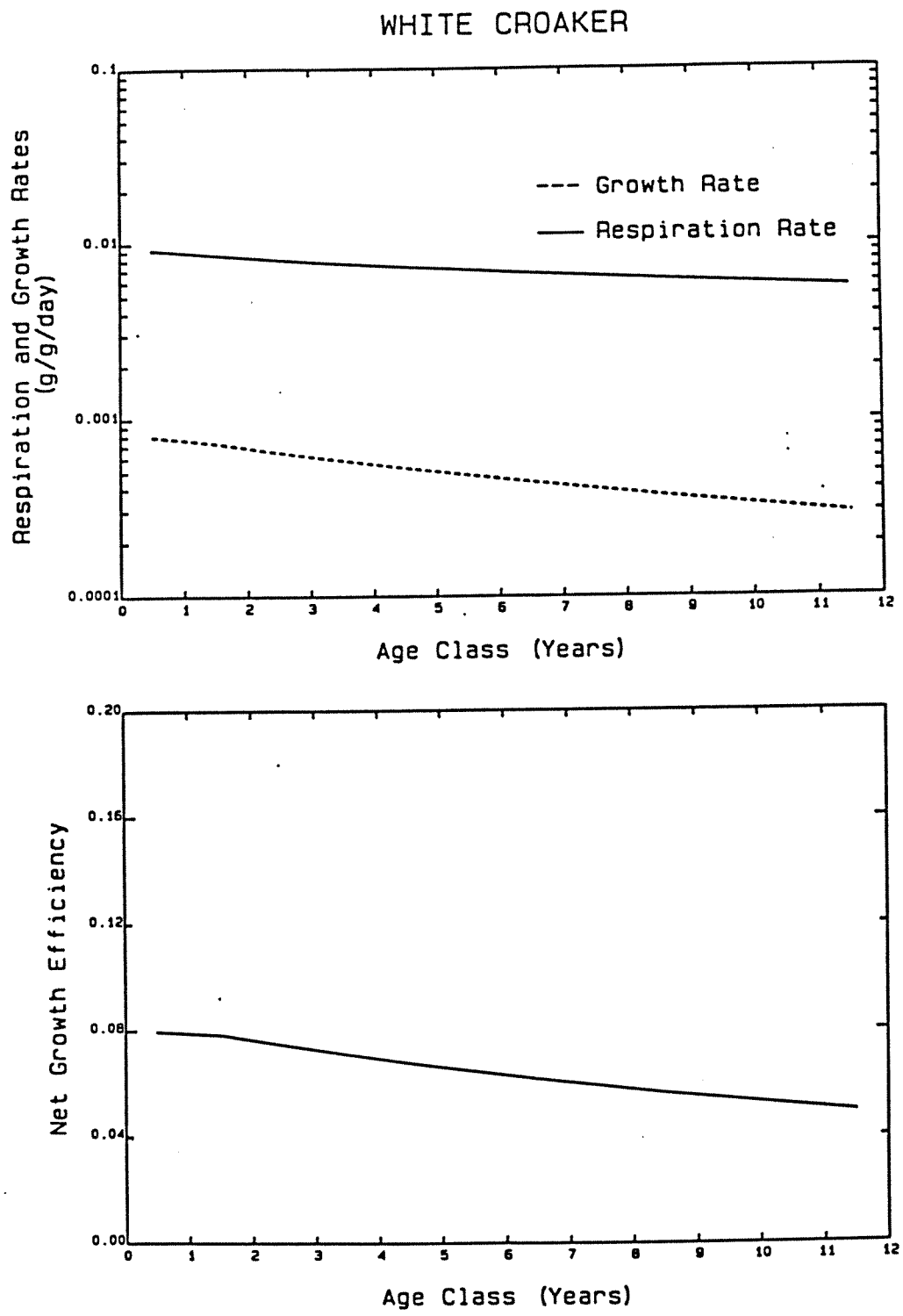
WHITE CROAKER

Figure 3-10. Calculated white croaker growth, respiration and net growth efficiencies as a function of age.
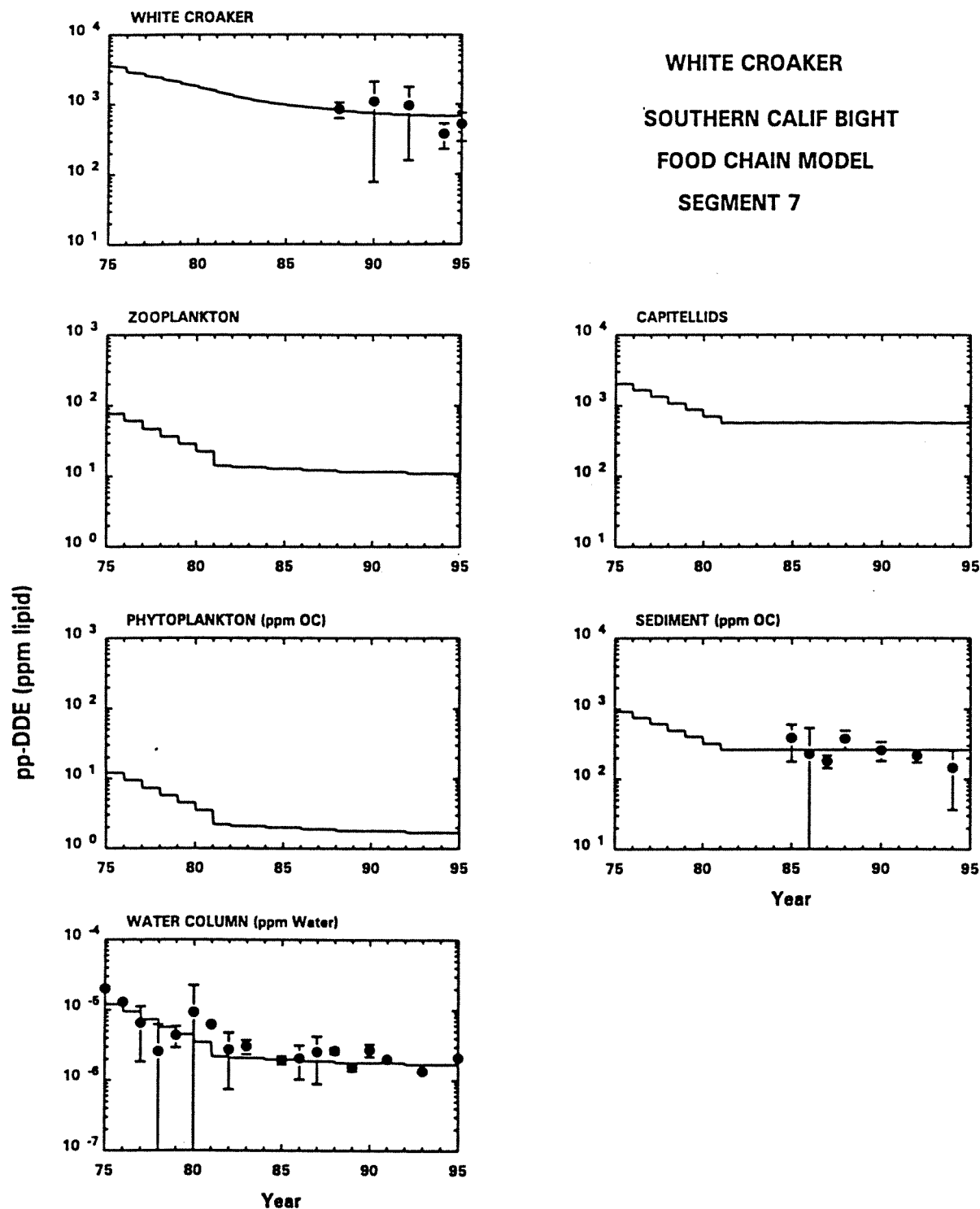
Figure 3-11. Computed lipid normalized p,p'DDE concentrations (lines) and data (symbols) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).
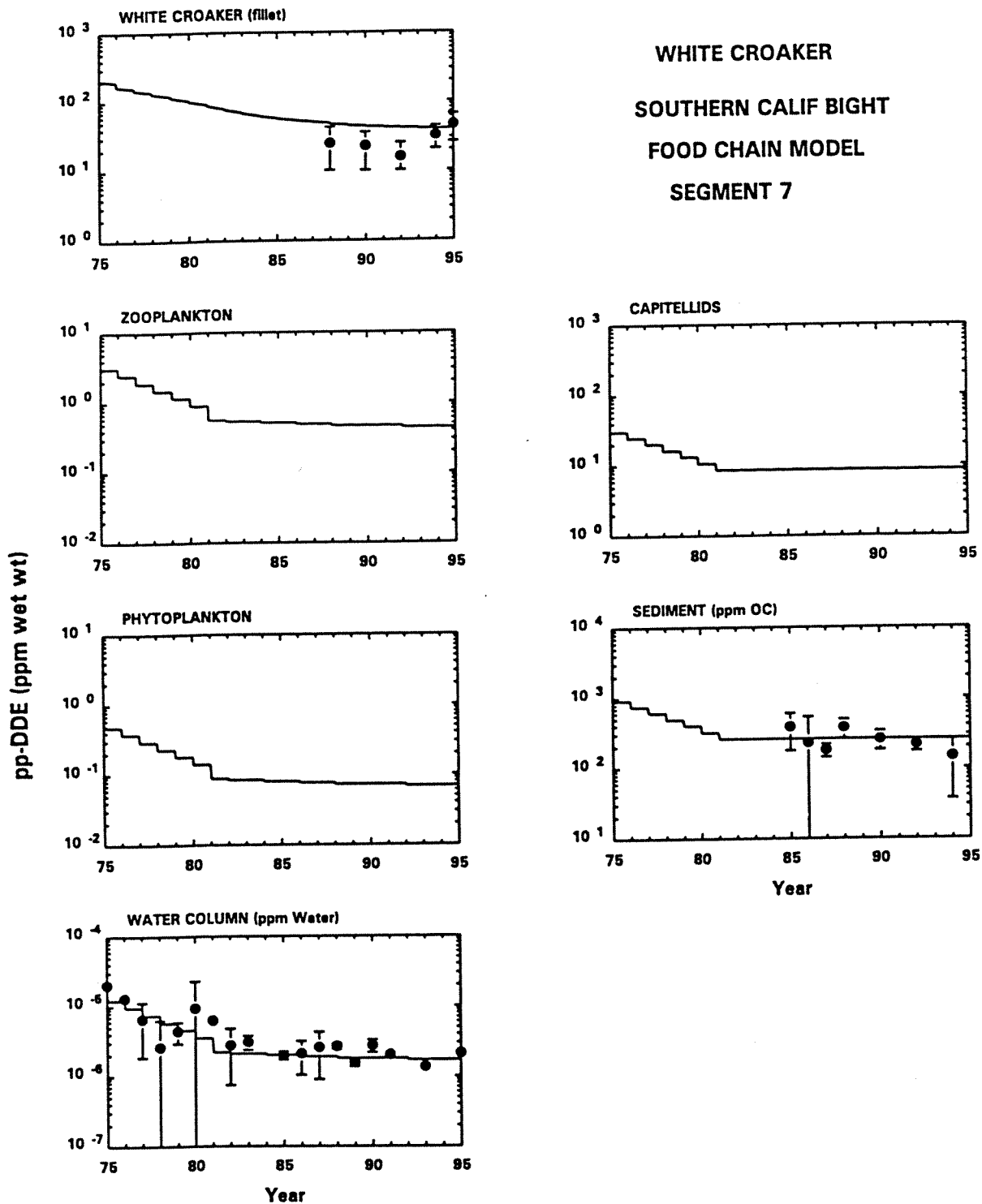
Figure 3-12. Computed wet weight p,p'DDE concentrations (lines) and data (symbols) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).

WHITE CROAKER

SOUTHERN CALIF BIGHT

FOOD CHAIN MODEL
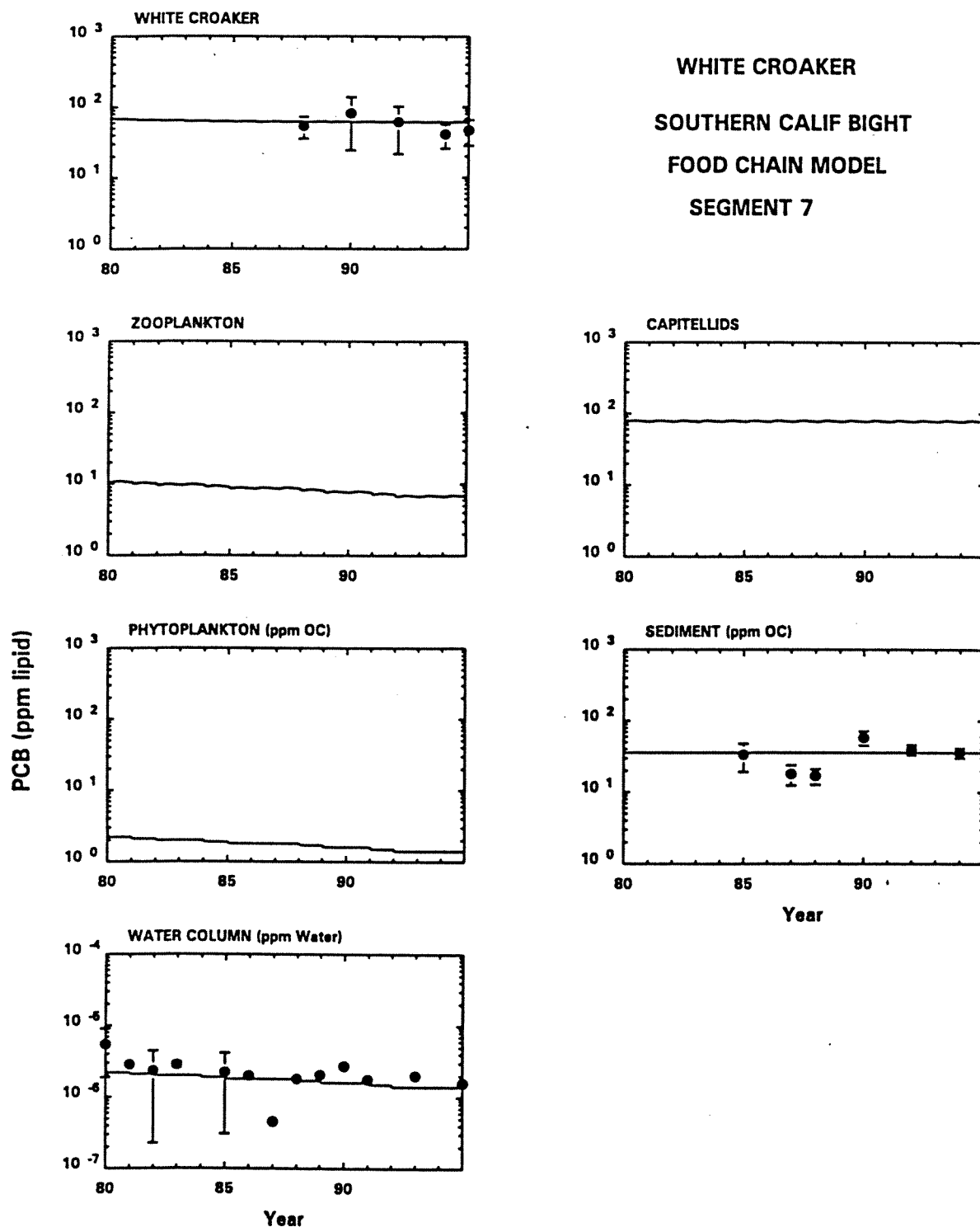
SEGMENT 7

Figure 3-13. Computed lipid normalized total PCB concentrations (lines) and data (symbols) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean.
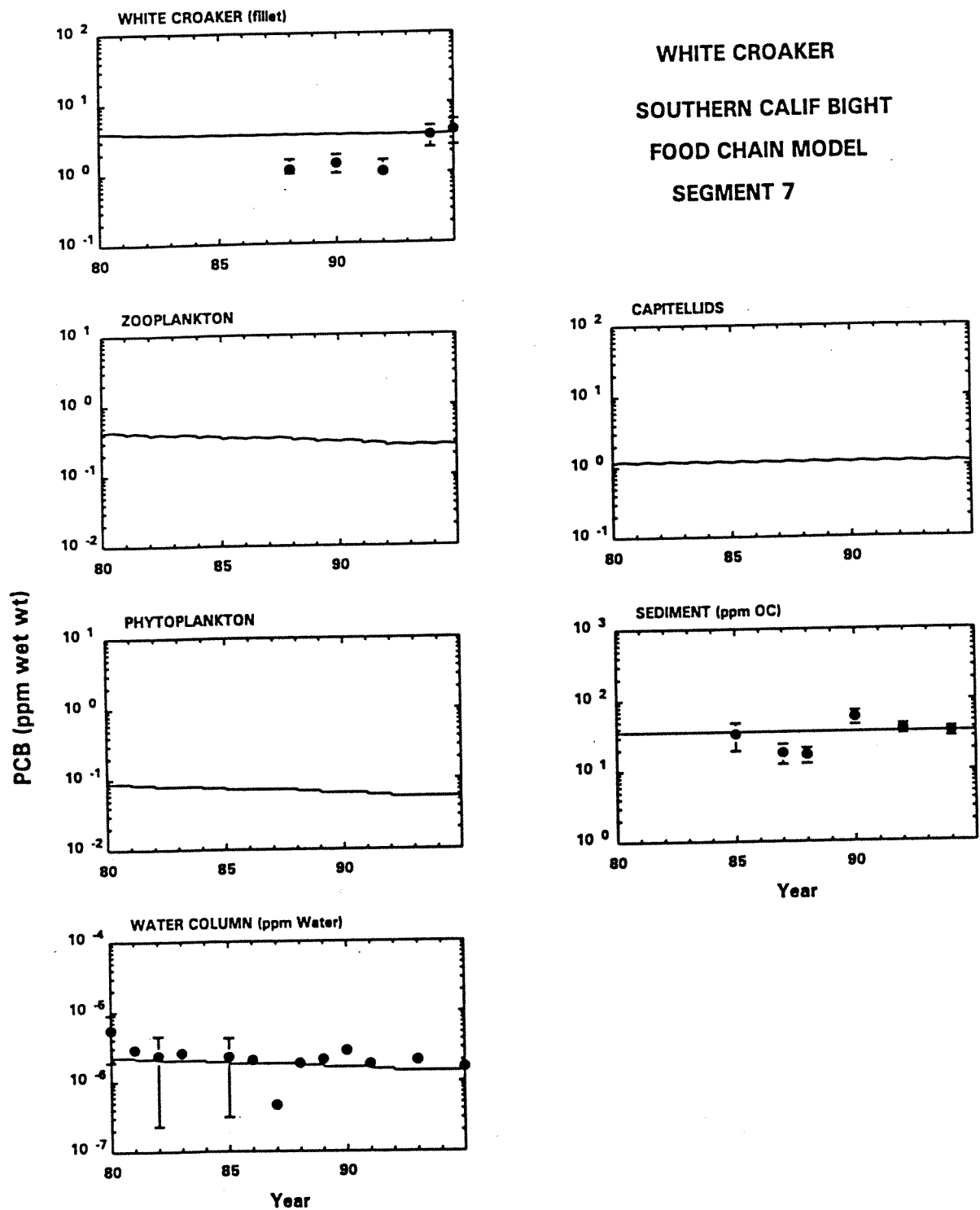
Figure 3-14. Computed wet weight total PCB concentrations (lines) and data (symbols) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean.

3.6   DOVER SOLE MODEL

3.6.1 Development of Species-Specific Physiological Parameters

3.6.1.1   Habitat

Dover sole eggs and larvae are pelagic, usually occurring in the top 50 meters of the water column. After approximately one to two years in the plankton, the larvae settle out in shallow nearshore waters (Horton 1989, Hagerman 1952, Hunter *et al.* 1990). The juveniles and adults prefer soft muddy or silty bottoms at depths between 35 to 180 meters and are often found concentrated in the vicinity of major municipal outfalls (Mearns & Harris 1975). As Dover sole mature, they are found in progressively deeper off shore waters (Hunter *et al.* 1990).

3.6.1.2   Food

The larvae feed on copepods and other planktonic organisms (U.S. Department of Commerce 1990). The immature and adults are selective benthic feeders who forage visually during the day. They prey primarily on polychaetes (Horton 1989).

For the model, Dover sole less then 1.5 years old feed on zooplankton. At age 1.5, Dover sole settle out of the plankton and begin their benthic existence, feeding on benthic invertebrates (Figure 3-15).

3.6.1.3   Movement and Migration

The location of Dover sole off the coast of California varies depending on stage of development and time of the year. After adult Dover sole spawn in deep waters of the continental slope, their eggs rise to the surface and become part of the plankton. As the larvae develop they remain in the plankton for one to two years before settling in shallower waters of the continental shelf (after reaching 35 to 65 mm). The juvenile fish remain inshore for a number of years as they mature into adults (individuals reach adulthood, spawning age, at 300 to 450 mm). As adults, Dover sole migrate annually to deeper offshore spawning grounds in the winter and to shallower inshore feeding grounds in the summer. As Dover sole age, the adults stay in deeper water during the summer months (Hagerman 1952, Hunter *et al.* 1990).
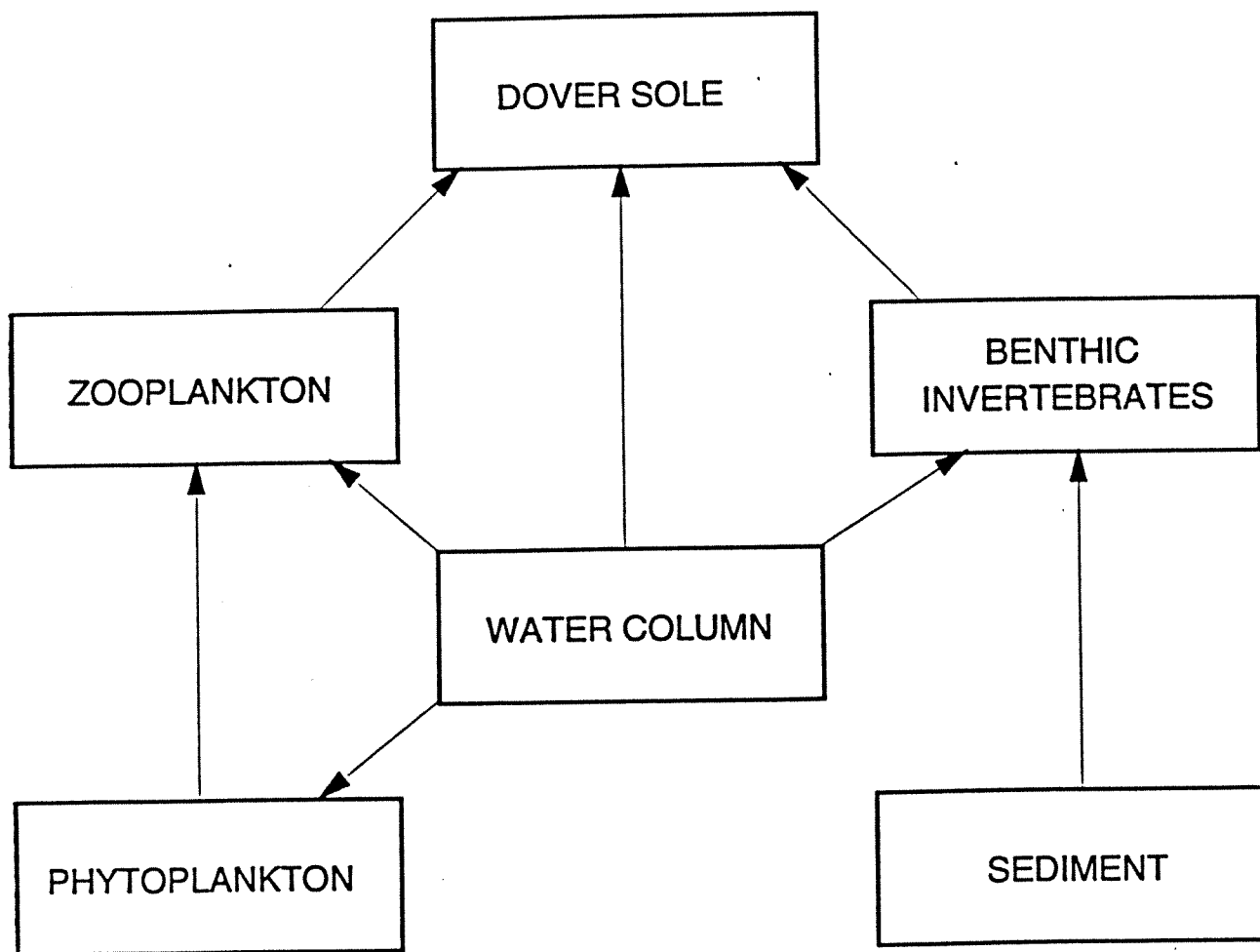
Figure 3-15. Dover sole food web.

The movement of adult Dover sole seems limited to onshore/offshore migration with little or no movement along the coast. However the larvae spend one to two years in the plankton during which time considerable along-shore dispersion is possible. In addition, the Dover sole population south of Point Conception is young, and is almost entirely immature (U.S. Department of Commerce 1990). Therefore, individuals from the southern California population are probably recruited from the planktonic larvae of more northern populations (U.S. Department of Commerce 1990).

Only young-of-the-year and immature fish were included in the model because of the lack of an adult Dover sole population off Palos Verdes. Migration was not included in the model, since juveniles do not move offshore as adults do.

3.6.1.4 Growth and Body Composition

The growth rate of Dover sole was computed in the same way as the growth rate of the white croaker. Length-age relationships for the Dover sole have been determined by Hagerman (1952) for fish collected in northern California and Oregon, by Demory (1975) for fish collected in Oregon, and by Mearns and Harris (1975) for fish collected from several California sites. Data from Hagerman (1952), Demory (1975) and from the Palos Verdes population sampled by Mearns and Harris (1975) are shown on Figure 3-16, top left panel. The values reported by Mearns and Harris (1975) indicate slower growth for Palos Verdes fish than for the fish collected in the more northern locations. Similarly, Mearns and Harris (1975) concluded that the fish of the Southern California Bight grew slower than northern California and Oregon fish.

The data used to compute growth rate and body composition are presented in Figure 3-16, which follows the same format as Figure 3-9. The relationship between weight and total length was determined by Mearns and Harris (1975), Demory (1975) and by Hunter (1992) for fish from central California (Figure 3-16, middle left panel). The length-age and weight-length relationships were combined to calculate age-weight relationships in order to estimate mass and mass growth rate for use in the model (Figure 3-16, bottom left panel). The data of Mearns and Harris (1975) suggest that after two years of age, fish from the Palos Verdes area grow more slowly than fish from more northerly locations.

The causes of the apparently slower growth of Dover sole collected on the Palos Verdes shelf in the early 1970's were not known, but may have included some effects of materials that

GROWTH INFORMATION

DOVER SOLE
IN THE SCB

Length/Age Relationship

Total Length (cm)

◇ Mearns 75, PVS
▽ Demory 75, OR
☐ Hagerman 52, NC-OR
— HQI Regression

Age (year)

Length Distribution

Length (cm)

Probability

Weight/Length Relationship

Weight (g)

◇ Mearns 75
▽ Demory 75
— Hunter 92, CC

Total Length (cm)

Percent Lipid/Length Relationship

Percent Lipid

Length (cm)

Weight/Age Relationship

Weight (g)

◇ Mearns 75
▽ Demory 75
— Hunter 92
& Hagerman 52

Age (year)

Percent Dry Weight/Length Relationship

Percent Dry Wt

Length (cm)

Figure 3-16. Dover sole growth and body composition data.

originated in the LACSD outfall (Mearns and Harris 1975). In addition, current growth rates are not known for Dover sole from the Palos Verdes area. Therefore, model runs were performed using both the faster growth rates for the more northerly fish as well as the slower growth rates from Palos Verdes shelf.

The percent lipid and percent dry weight for the fish included in the study are relatively constant over the size range of fish included in the study: 18 percent for dry weight and 2.2 percent for lipid for muscle (fillet; Figure 3-16). The average percent dry weight (18 percent) agrees well with the number reported by Hunter *et al.* (1990) for fish of similar size (17.1 percent). Because the food chain model requires values on a whole-body basis, muscle lipid values were converted to whole-body values by multiplying by 5.5 (Table 3-4), and muscle percent dry weight was assumed to represent the whole-body values.

3.6.1.5   Respiration

Respiration coefficients are available for another flatfish, the winter flounder (Voyer and Morrison (1971). These coefficients were used in a model of bioaccumulation of PCBs by flounder in New Bedford Harbor, Massachusetts (HydroQual 1990; Connolly 1991) and are used here. Figure 3-17 details the relationships between growth, respiration and net growth efficiency as a function of age for Dover sole.

3.6.2 Model Results

The model calculations were performed for the periods 1975 to 1995 (p,p'DDE) and 1980 to 1995 (PCBs). The model calculated levels of p,p'DDE that are within the range of the annual average data values (Figures 3-18 and 3-19). Calculated PCB levels are within the range of lipid-based PCB averages and slightly higher than the wet-weight-based averages (Figures 3-20 and 3-21). Thus, results indicate that the contamination in Dover sole that are caught in the area of the outfall originated in local contaminated sediments.

The results described above were calculated using the faster growth rate for Dover sole (see discussion above). Using the slower growth rate that was measured in Palos Verdes populations in the early 1970's, the calculated body burdens are higher by approximately 20 percent. This does not change the conclusions qualitatively.
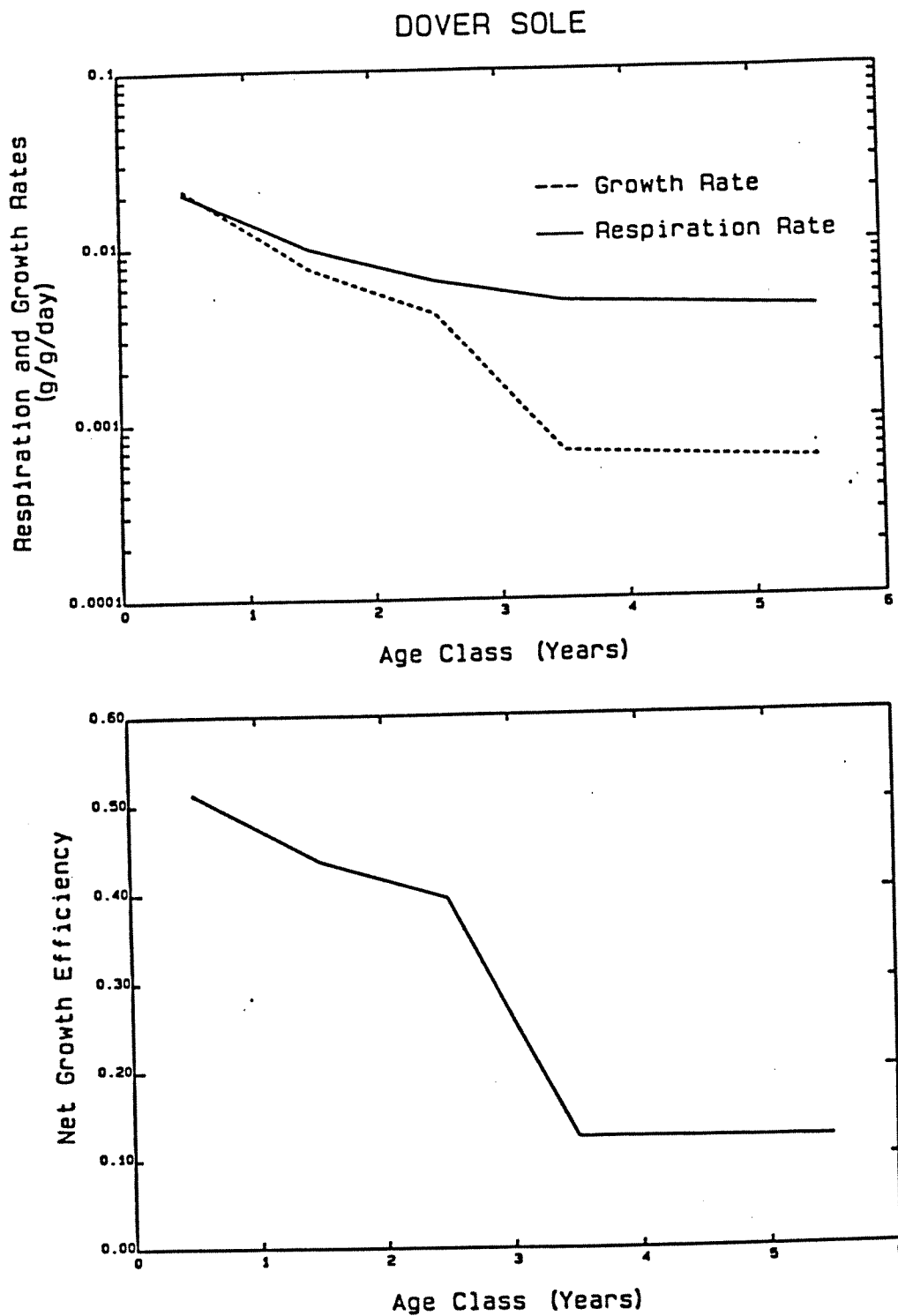
DOVER SOLE

Figure 3-17. Calculated Dover sole growth, respiration and net growth efficiencies as a function of age.

Figure 3-18. Computed lipid normalized p,p'DDE concentrations (lines) and data (symbols) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).

Figure 3-19. Computed wet weight p,p'DDE concentrations (lines) and data (symbols) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).
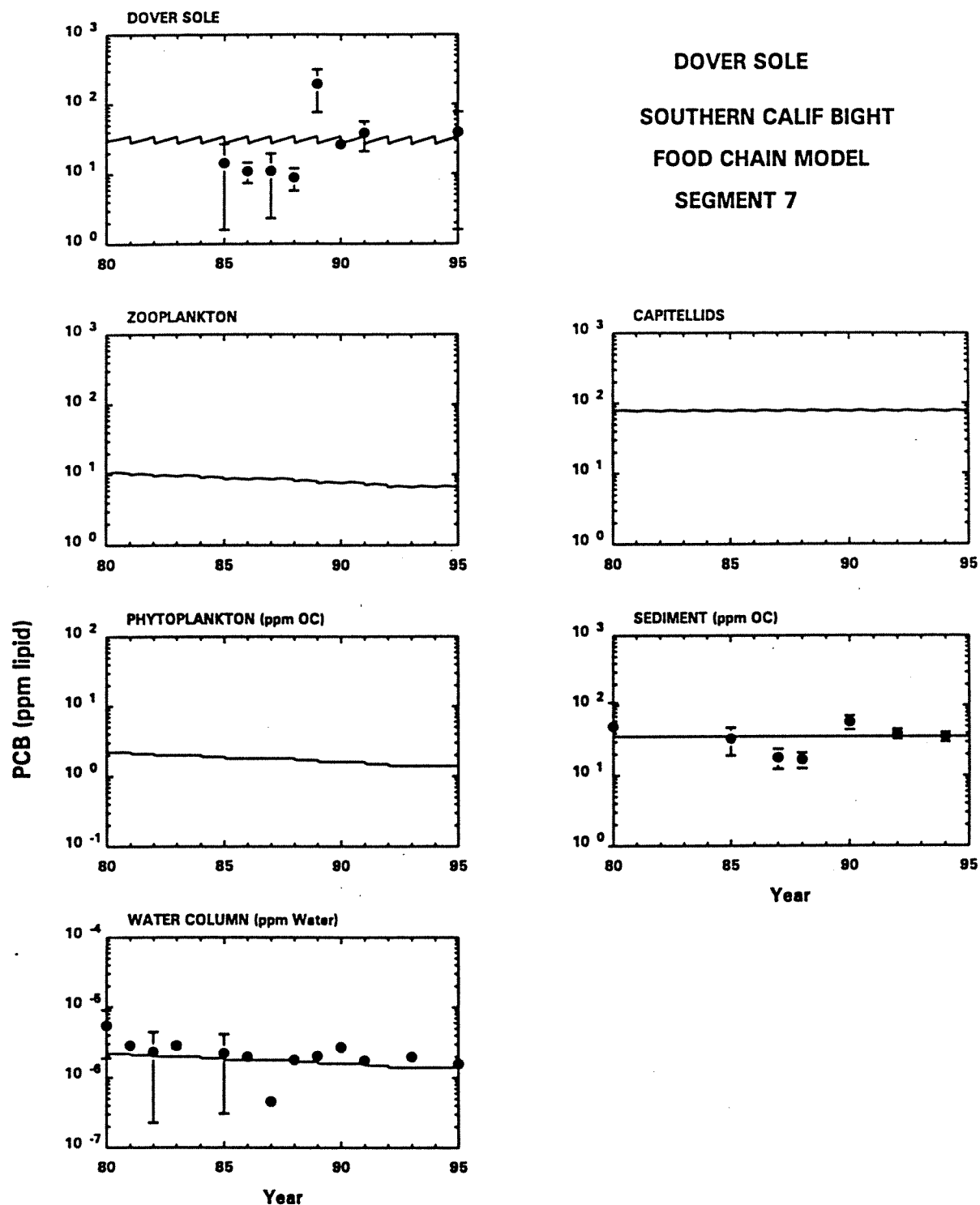
Figure 3-20. Computed lipid normalized total PCB concentrations (lines) and data (symbols) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).

DOVER SOLE (fillet)

ZOOPLANKTON

PHYTOPLANKTON

WATER COLUMN (ppm Water)

DOVER SOLE

SOUTHERN CALIF BIGHT

FOOD CHAIN MODEL

SEGMENT 7

CAPITELLIDS

SEDIMENT (ppm OC)

PCB (ppm wet wt)

Year

Year

Figure 3-21. Computed wet weight total PCB concentrations (lines) and data (symbols) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7; arithmetic means +/- 2 standard errors of the mean).

The model results for the Dover sole exhibit greater variation within each year than do the white croaker. This is so because the results of the Dover sole simulations are presented for age 2 to 3 fish, and these fish have been feeding in the sediments for only 1/2 year prior to their second birthday. Thus, during their second and third year of life, their body burdens are increasing rapidly as they ingest highly contaminated sediments. In contrast, the white croaker results are presented for older fish that have been feeding on the bottom for several years.

3.7 KELP BASS MODEL

3.7.1 Development of Species-Specific Physiological Parameters

3.7.1.1 Habitat

Kelp bass are an important sport fish found in areas of physical relief such as rocky reefs or kelp beds off the coast of southern California (Hobson and Chess 1986). Juveniles generally inhabit the surf zone and inshore kelp beds and are found from the surface to depths of about 30 meters. Adult kelp bass prefer the clearer waters on the outer margins of kelp beds and are usually found in waters less than 55 meters deep (NOAA Marine Atlas 1990). However, they are known to exist in rocky or sandy areas in the absence of kelp. In any case, they are generally associated with structures of some sort, for example, oil platforms (Love 1996).

3.7.1.2 Food

During the first three or four months of life kelp bass feed in the kelp canopy primarily on planktonic organisms such as copepods, shrimp and gammarid amphipods. After these first few months, kelp bass feed closer to the bottom. By about two years of age (about 150 mm standard length (SL)), kelp bass become mostly piscivorous, consuming fish and cephalopods (Quast 1968). In addition, invertebrates and crustaceans constitute a component of the adult kelp bass diet (Quast 1968).

Kelp bass diet can vary in space and time. Kelp bass are switch-feeders, having the ability to feed on both small and large prey. As switch-feeders kelp bass readily consume the more abundant prey, therefore the composition of the diet depends to some degree on the composition of the kelp bed community. Kelp bass from different areas show different diets (Love and Ebeling 1978). In addition, there is seasonal variation in kelp bass dietary

preferences. Feeding peaks during the fall and late spring and this may be related to reproductive cycles (Quast 1968).

In the model, two components to the kelp bass food web are considered. They include a macroalgae-based component and a phytoplankton-based component. The macroalgae component consists of macroalgae - invertebrates - fish - kelp bass. The macroalgae component represents kelp as well as other plant material such as algae growing on animate or inanimate surfaces. The phytoplankton based component consists of phytoplankton - zooplankton - fish - kelp bass. Model simulations were performed using a combination of the macroalgae- and phytoplankton-based components (Foster & Schiel 1985). Juvenile kelp bass (ages 0 and 1) consumed 50% invertebrates that feed on macroalgae and 50% zooplankton. Adult kelp bass (ages 2 through 11) consumed 75 percent kelp bed fish and 25 percent invertebrates that in turn feed on macroalgae. Kelp bed fish feed on kelp bed/hard bottom invertebrates (Figure 3-22).

Thus, the food web is considered pelagic. This conclusion is supported by examination of the relationship between contaminant levels in the kelp bass, the mussels and the sediments. Risebrough (1987) collected matched sediment, mussel and kelp bass data from several widely spaced locations with very different contamination levels. The relationships between kelp bass and mussel contaminant levels, and between kelp bass and sediment levels are shown on Figure 3-23. One strength of this analysis is that all measurements were made in a single study, thus reducing analytical uncertainty. Fraction organic carbon was not available for this data set, so a constant value of 0.01 was used for the entire study area. More realistic, site-specific values would likely increase the strength of the following conclusions:

- Kelp bass contaminant levels are roughly proportional (slope = 0.7) to mussel (and therefore water column) contaminant levels (Figure 3-23, bottom panel). Kelp bass contaminant levels are correlated with sediment levels, but the slope is less than one (0.3; Figure 3-23, top panel). This is more consistent with a food web based in the water column than one based in the sediment. Note that even if the fraction organic carbon was ten-fold greater in the Palos Verdes shelf sample, this conclusion would still hold.

- In the area of highest contamination (Palos Verdes shelf), the kelp bass concentration (ug/g lipid) is less than the sediment concentration (ug/g organic carbon; Figure 3-23, top panel). This is not consistent with bioaccumulation through a benthic food web.
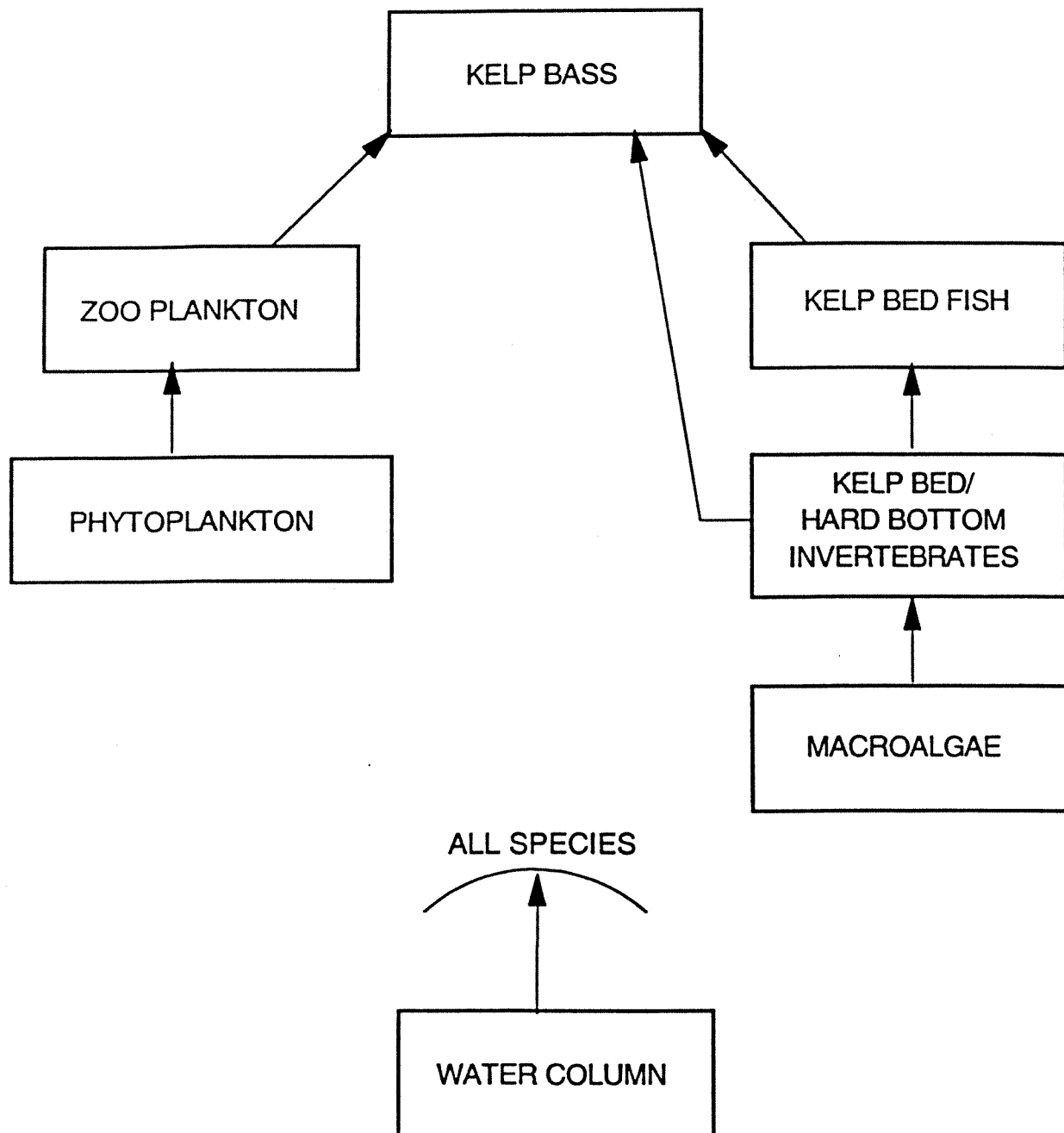
Figure 3-22. Kelp bass food web.
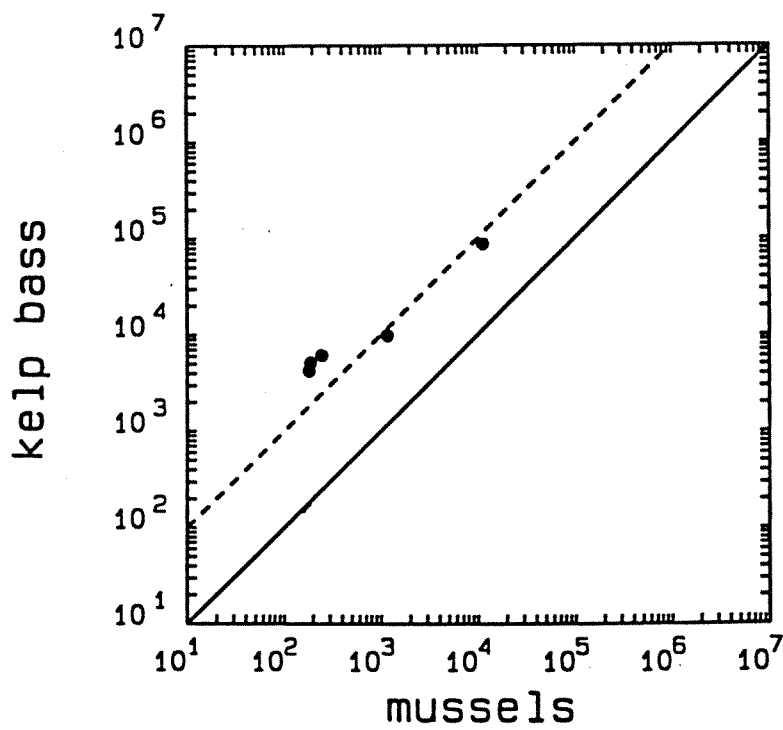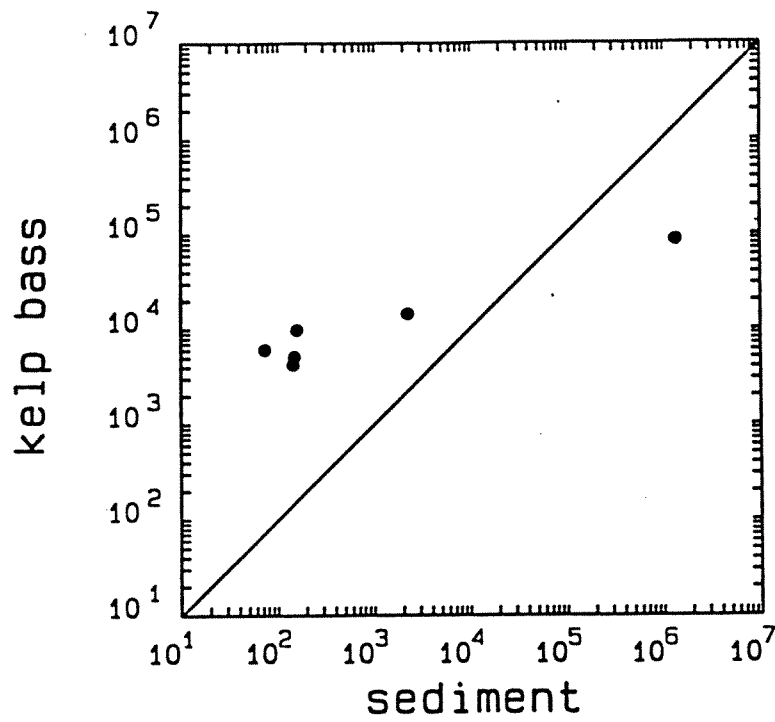
Figure 3-23. Comparison between kelp bass, mussels and surface sediment DDE concentrations.

- Kelp bass levels (ug/g lipid) are approximately ten-fold greater than levels in the mussels (Figure 3-23, bottom panel). This is consistent with bioaccumulation through a four-step food chain consisting of macroalgae, invertebrates, small fish, and kelp bass.

The model output for the kelp bed/hard bottom invertebrates was compared to yellow rock crab data (Figures 3-26 and 3-27). Note that yellow rock crab may not be a an important prey of kelp bass therefore its use is for comparative purposes only.

3.7.1.3   Movement and Migration

Kelp bass are perceived as a relatively sedentary species (U.S. Department of Commerce 1990). For example, a tagging study conducted in southern California in the 1950's by the California Department of Fish and Game showed relatively little movement of kelp bass (Love 1996). However, anecdotal information suggests kelp bass may be more mobile than previously suspected. For example, the kelp bass population at Naples Reef, a rocky reef near a large kelp bed off Santa Barbara, has been estimated to be about 400 individuals. However, over the course of one year, over 5,000 kelp bass were caught there by a single sport fishing vessel. Another report indicates skippers of area sport fishing vessels observe teams of kelp bass moving onto offshore reefs during certain periods, particularly spawning season. One possible explanation is that kelp bass living in optimal environments move little, while those living in suboptimal environments may seek better conditions (Love 1986, unpublished data).

Contaminant data from the HydroQual database were examined to help determine the extent of kelp bass movement along the Palos Verdes Shelf. First, the value of the slope of DDE concentrations in kelp bass versus mussels is near unity (Figure 3-23; data of Risebrough 1987). If kelp bass migrated freely throughout the area, then there would be no relationship between kelp bass levels and mussel levels. In addition, concentrations of p,p'DDE in kelp bass measured in other studies exhibit an along-shore gradient, decreasing away from the outfall (Figure 2-13). This is consistent with limited along-shore movement.

Therefore, kelp bass movement is probably limited. To account for the uncertainty of kelp bass movement, two model simulations were performed. The first considered kelp bass as a non-migratory fish spending all its time on the Palos Verdes Shelf. The second allowed adult

kelp bass (greater than about 150 mm SL) to move from the higher contaminated Palos Verdes Shelf to the lesser contaminated Santa Catalina Island for three months of the year.

3.7.1.4 Growth and Composition

A relationship between length and age in kelp bass determined for fish in the southern California area was used (Quast 1968; Figure 3-24, top left panel). This same study included a relationship between length and weight (Figure 3-24, center left panel). The relationships are as follows:

for 65 - 245 mm SL,   log W (grams) = 3.256 log L (mm SL) - 5.178
for 255 - 615 mm SL,   log W (grams) = 2.725 log L (mm SL) - 3.913.

Using these two relationships, a correlation between weight and age was determined (Figure 3-24, bottom left panel). This last relationship was used to calculate growth rates (g/g/day).

The mean lipid and dry weight fractions of kelp bass muscle tissue in the database are about 0.024 (g lipid/g wet weight; Figure 3-24, center right panel) and 0.23 (g dry weight/g wet weight; Figure 3-24, bottom right panel) respectively. Two outliers, lipid values for 3 and 11 year old kelp bass ,were not included in the average. Muscle lipid fraction was converted to total lipid fraction using a whole-body:muscle lipid ratio of 2.6. Lipid and dry weight fractions of the kelp bed fish and pelagic fish were assumed the same as those of kelp bass.

The kelp bass in the database range from 250 to 350 mm standard length (2 fish are about 500 mm SL). This range corresponds to ages three to five years. To compare model output with this data, contaminant levels in the model from these age classes were used.

3.7.1.5 Respiration

Species-specific respiration coefficients were not available for kelp bass. A general relationship was used to empirically describe resting respiration rate as a function of weight and temperature (Hemmingsen 1960). Resting metabolic rate was multiplied by 2 to give the active rate (Peters 1983). The respiration rate decreased exponentially from about 0.021 g/g/day (grams oxygen/gram body weight/day) for age 1 kelp bass to about 0.003 g/g/day for 12 year olds (Figure 3-25). The growth rate of 1 year old kelp bass was approximately 0.015 g/g/day

GROWTH INFORMATION

Length/Age Relationship

Quast (1971)

KELP BASS
IN THE SCB

Length Distribution

N=299

Weight/Length Relationship

Quast (1971)

Percent Lipid/Length Relationship

Weight/Age Relationship

Percent Dry Weight/Length Relationship
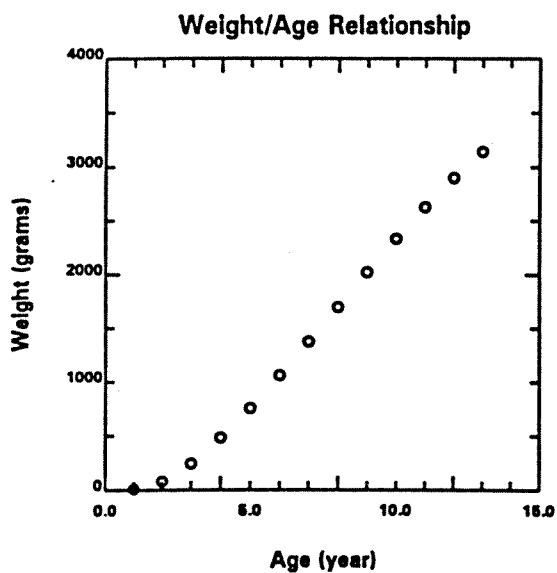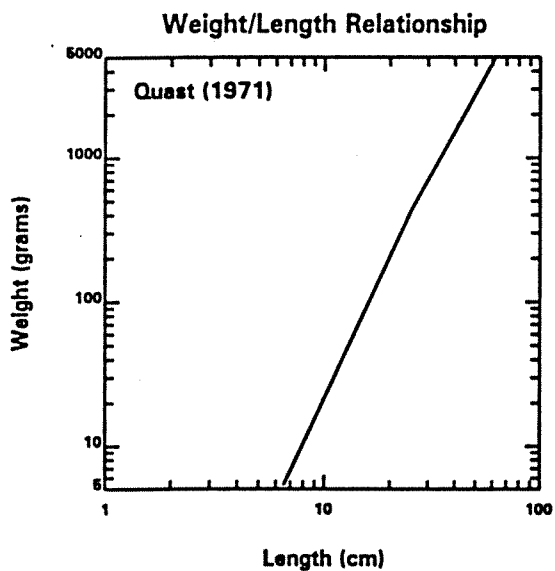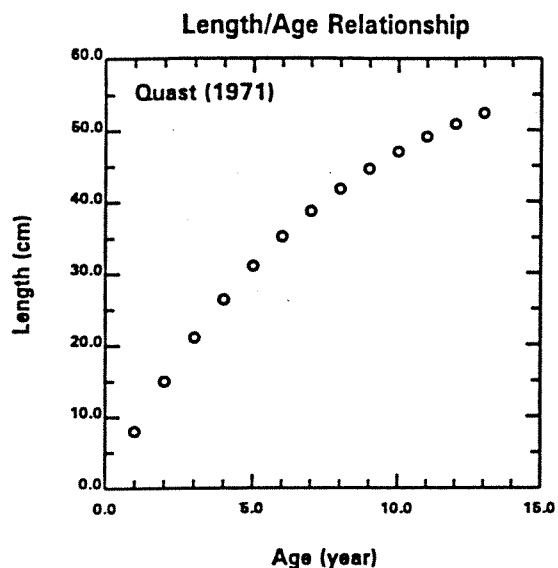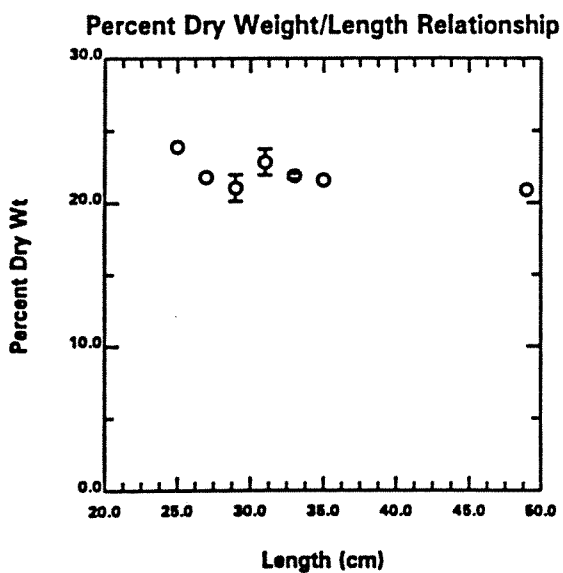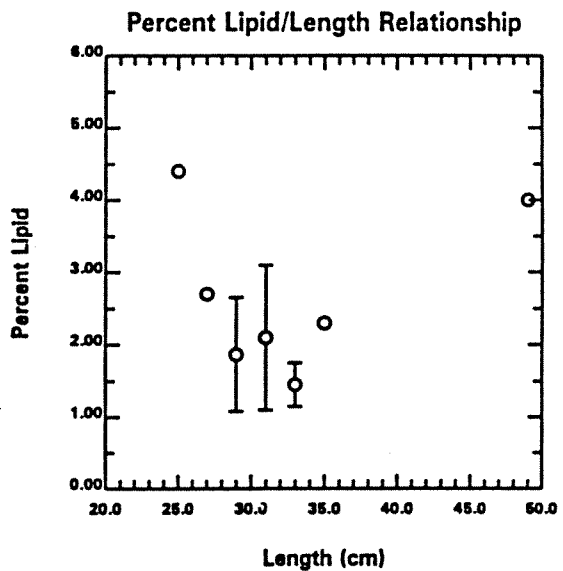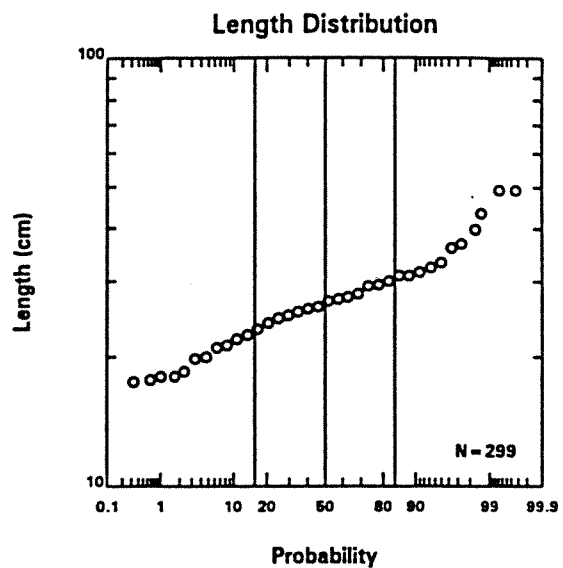
Figure 3-24. Kelp bass growth and body composition data.
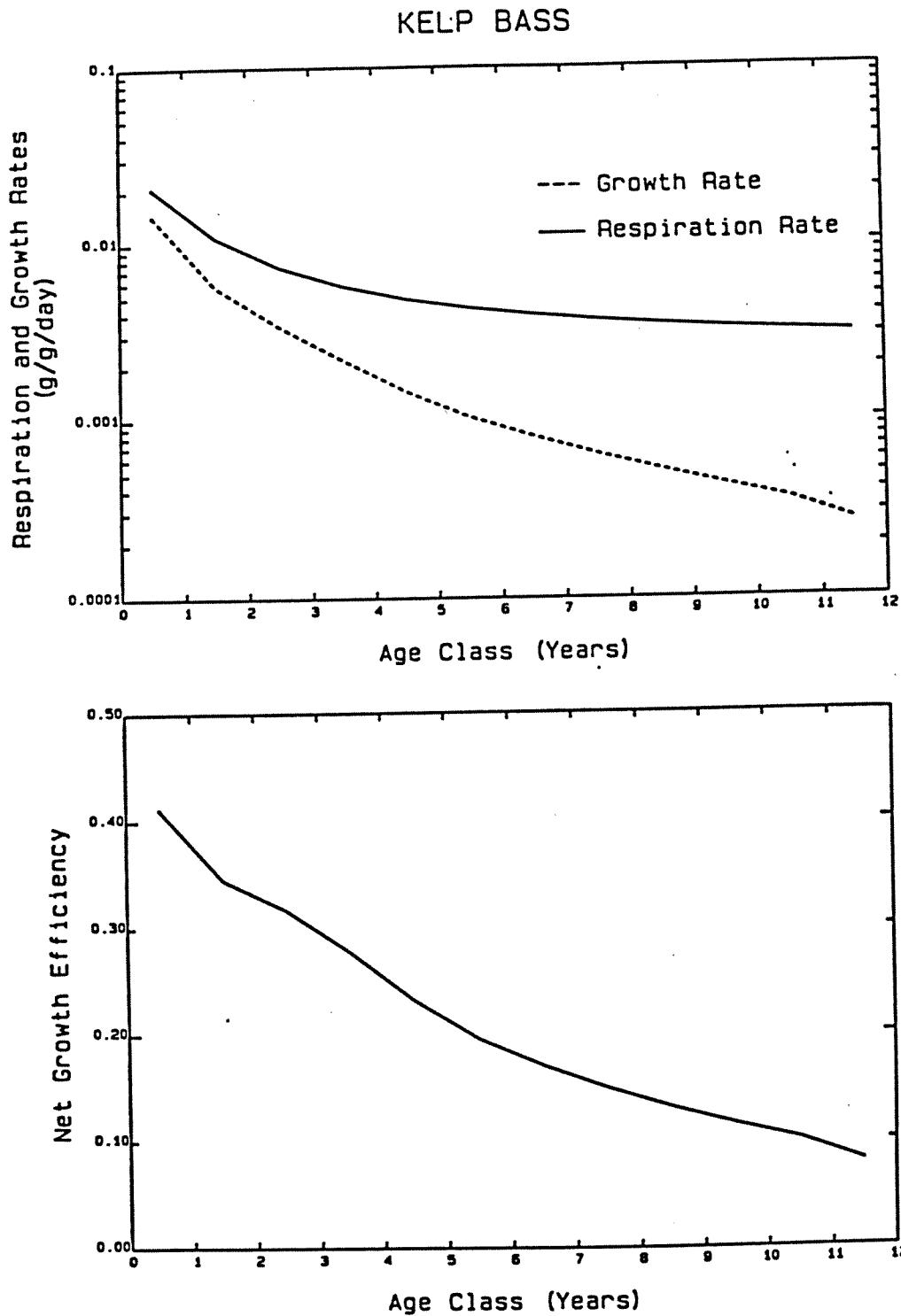
Figure 3-25. Calculated kelp bass growth, respiration and net growth efficiencies as a function of age.

and declined to near zero for 12 year olds. Net growth efficiency was used as a check on the reasonableness of the respiration and growth rates. As typically observed, the net growth efficiency decreased with age from about 0.41 to about 0.08 (Figure 3-25). The bioenergetics of the pelagic fish and the kelp bed fish were calculated in the same manner as for the kelp bass.

3.7.2 Model Results

Two model simulations were performed. The first simulation considered kelp bass as a non-migratory species, spending its entire life over the highly contaminated Palos Verdes Shelf. The second simulation allowed adult kelp bass (ages two and older) to migrate to the lesser contaminated Santa Catalina Island for three months starting in early June. Both simulations were performed for p,p'DDE and Total PCBs, and results are presented on wet-weight and lipid-normalized bases.

Lipid-normalized results of the first simulation (no migration) for p,p'DDE are presented in Figure 3-26. Model predictions are similar to the available kelp bass data. Calculated concentrations decrease more than an order of magnitude from 1970 to about 1983 and decline less dramatically thereafter. This pattern is a direct result of the temporal pattern exhibited by the mussels (and by extension, the dissolved water column concentrations). On a wet-weight basis, predicted contaminant levels are slightly high during the early 1970's but are similar to kelp bass concentrations measured in the 1980's (Figure 3-27). Model predictions for PCBs are also similar to values measured since 1980 on both wet-weight and lipid-normalized bases (Figures 3-28 and 3-29).

The p,p'DDE and PCB data for 1987 on Figures 3-26 through 3-29 were collected by Pollock *et al.* (1991). Data for all other years were collected by LACSD. On a lipid basis, the Pollock *et al.* (1991) values are similar to the data collected by LACSD (Figures 3-26 and 3-28). However, on a wet-weight basis, the Pollock values are considerably lower than the LACSD values (Figures 3-27 and 3-29). This pattern, namely, that the data of Pollock *et al.* (1991) are low relative to other studies on a wet-weight basis but similar on a lipid basis, is also observed in other species and may reflect differences in the extraction techniques (see Pollock *et al.* 1991). All data are included in the plots for comparison.

The wet weight-based p,p'DDE data for yellow rock crab are similar to the model predictions (Figure 3-27). On a lipid basis, the model calculates values lower than the data.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

p,p - DDE

ppDDE Concentration (ppm lipid)
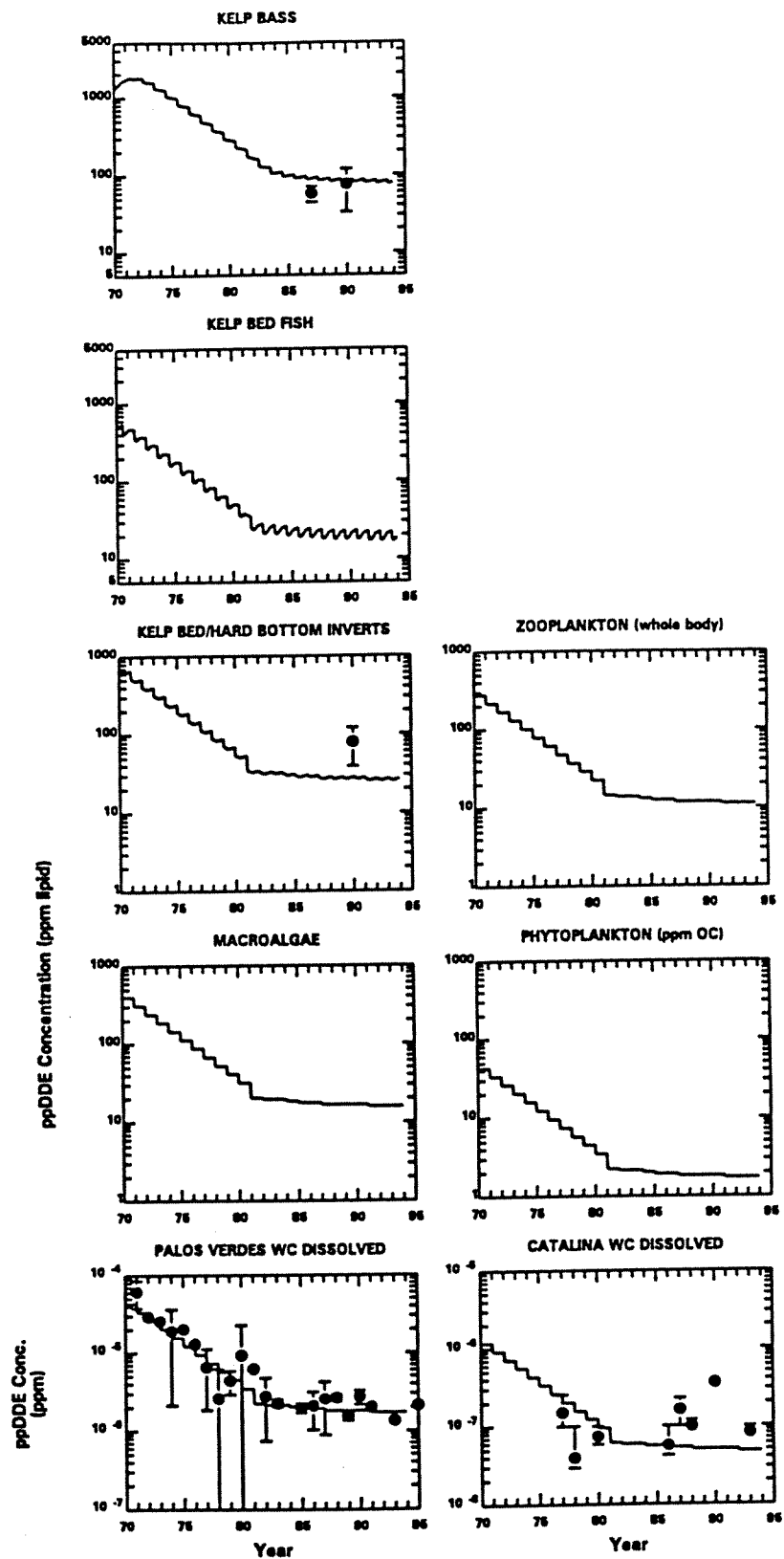
ppDDE Conc. (ppm)

Figure 3-26. Computed lipid normalized p,p'DDE concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario. Data are arithmetic means +/- 2 standard errors of the mean.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE
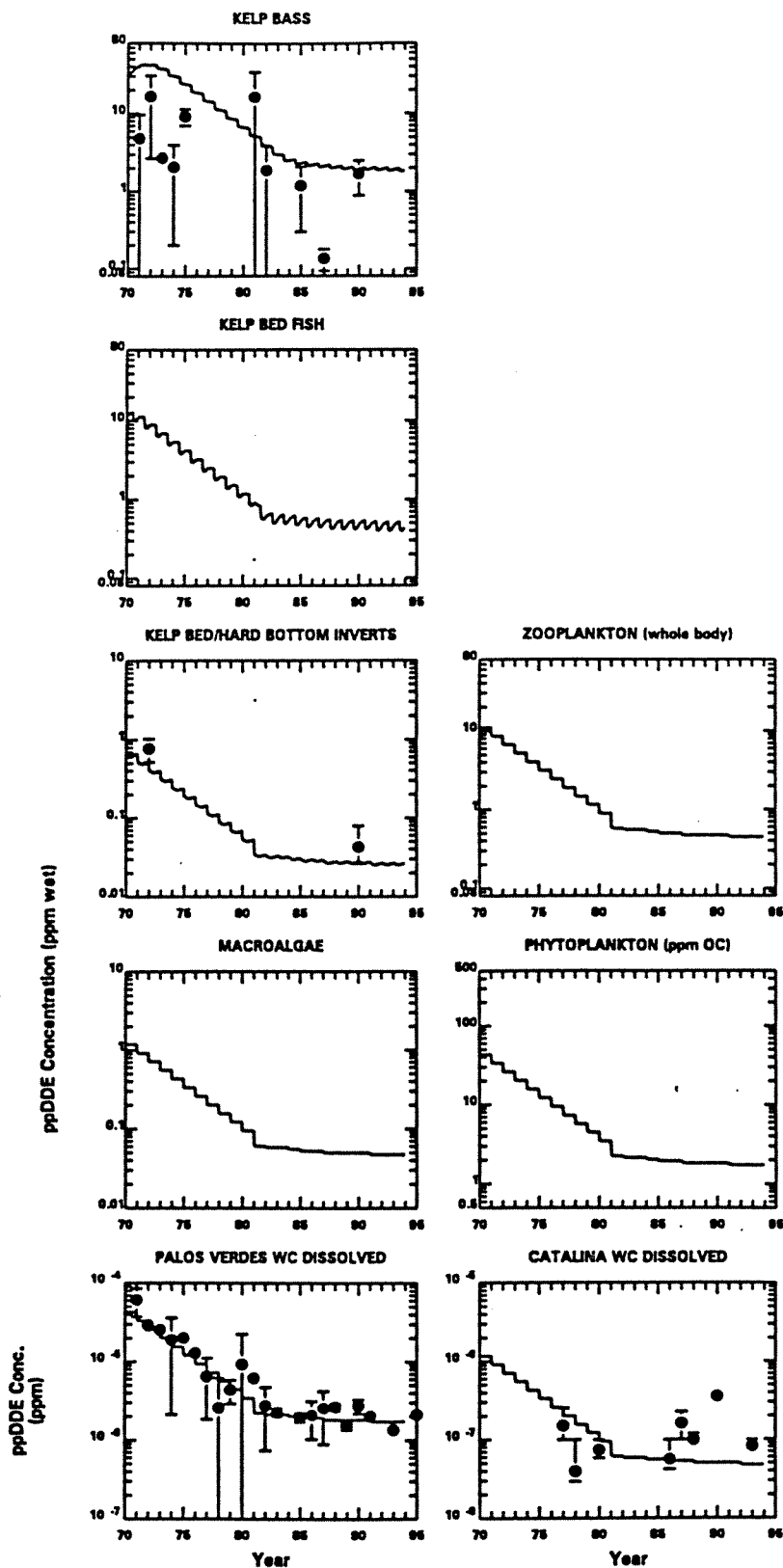
p,p - DDE

Figure 3-27. Computed wet weight p,p'DDE concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario. Data are arithmetic means +/- 2 standard errors of the mean.

KELP BASS

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

TOTAL PCBs

KELP BED FISH

KELP BED/HARD BOTTOM INVERTS

ZOOPLANKTON (whole body)

MACROALGAE

PHYTOPLANKTON (ppm OC)

PALOS VERDES WC DISSOLVED

CATALINA WC DISSOLVED

tPCB Concentration (ppm lipid)

tPCB Conc. (ppm)

Year

Year
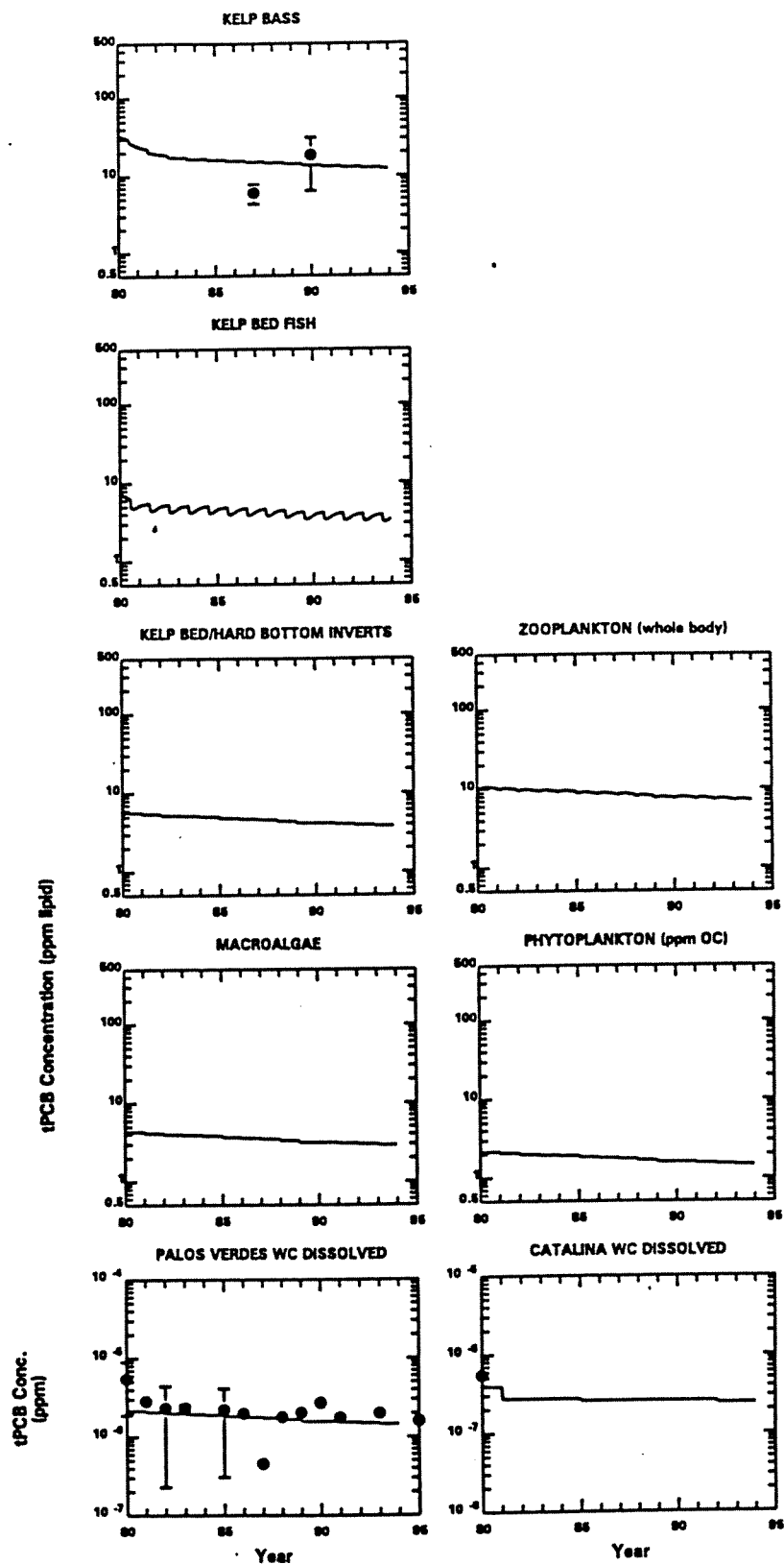
Figure 3-28. Computed lipid normalized total PCB concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario. Data are arithmetic means +/- 2 standard errors of the mean.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE
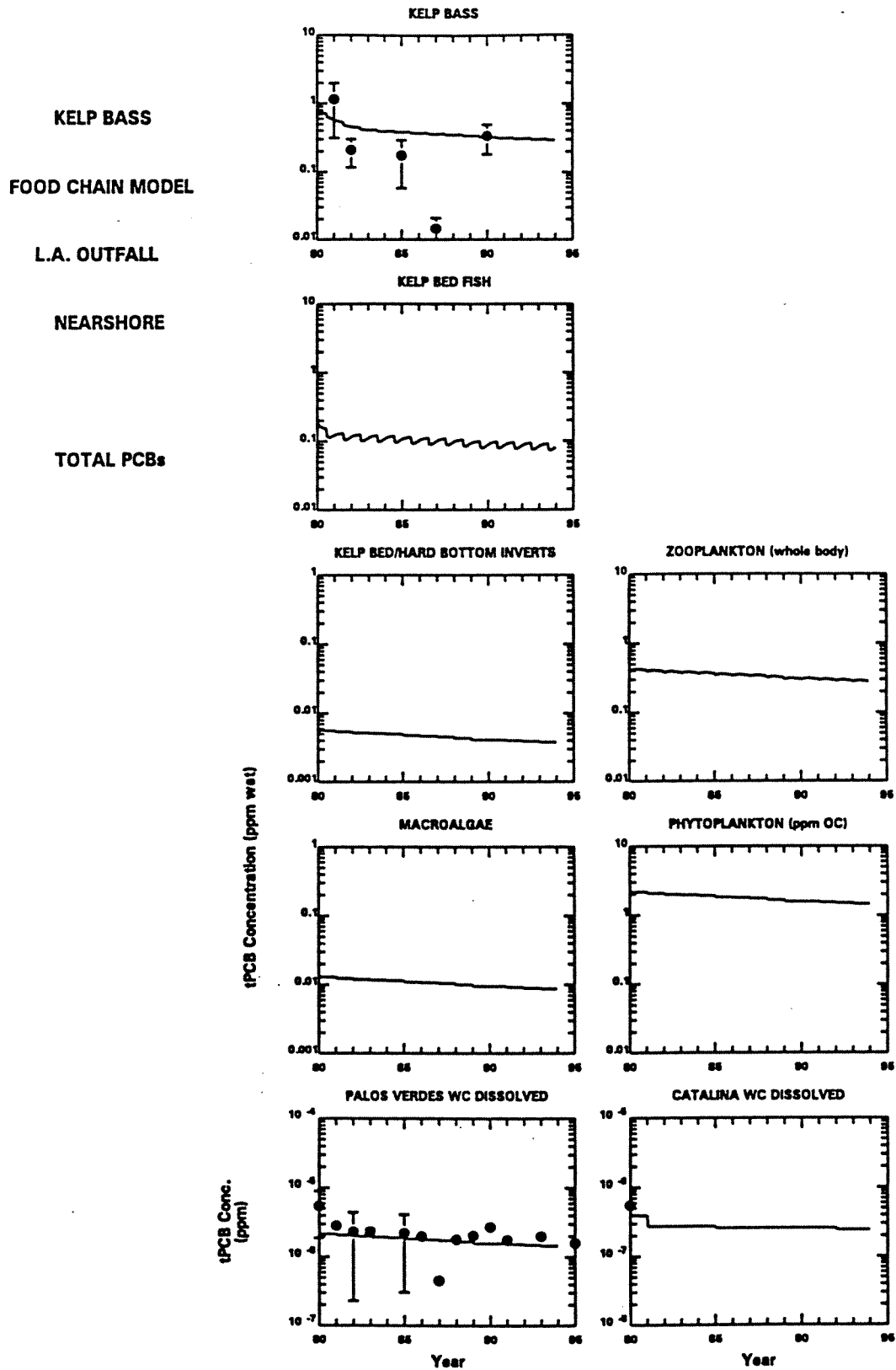
TOTAL PCBs

Figure 3-29. Computed wet weight total PCB concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario. Data are arithmetic means +/- 2 standard errors of the mean.

This is because the lipid content of the yellow rock crab data (0.003) is lower than the lipid content used in the model (0.015, based on data for lobster from New Bedford Harbor, Massachusetts). The latter value was considered a more realistic estimate for potential invertebrate prey species in general (Hepher 1988).

The inclusion of adult migration resulted in a 15 to 20 percent decrease in p,p'DDE and total PCB concentrations. These results are presented in Figures 3-30 through 3-33. If the fish spend all of their time near Santa Catalina Island, then their calculated levels are more than 10-fold lower than the first simulation for p,p'DDE or about 5-fold lower for PCB (results not shown).

Thus, model results suggest that contamination from local Palos Verdes Shelf waters is necessary and sufficient to produce the contaminant levels observed in kelp bass caught on the shelf. These fish could not attain the observed levels if they fed primarily in areas away from the shelf. If they fed only part of the time away from the shelf, then the primary source of contaminant would still be the shelf.

3.8   VALIDATION OF FOOD WEBS MODELS

To test the validity of the food webs models, simulations of p,p'DDE in each species were performed for the region extending from the lower Santa Monica Bay to the Los Angeles Harbor. This region was selected because of the large spatial gradients observed in the fish, mussel and sediment data. For each species, independent simulations were conducted for eight zones within this region (a total of 24 model runs). Segment-specific exposure concentrations were directly input for each model run.

Steady-state concentrations in the fish were computed using zone average sediment and water column concentrations for the period from 1985 to 1995. The steady-state assumption is reasonable because concentrations of p,p'DDE in the sediments and water column show no temporal trends after 1985. Model results were then compared to p,p'DDE levels observed in fish collected after 1985.

Computed and observed lipid-based p,p'DDE levels in white croaker, Dover sole and kelp bass are presented in Figure 3-34. Wet weight-based values are presented in Figure 3-35. These figures do not include the kelp bass data of Pollock (1991) (see section 3.7.2). For

KELP BASS

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE
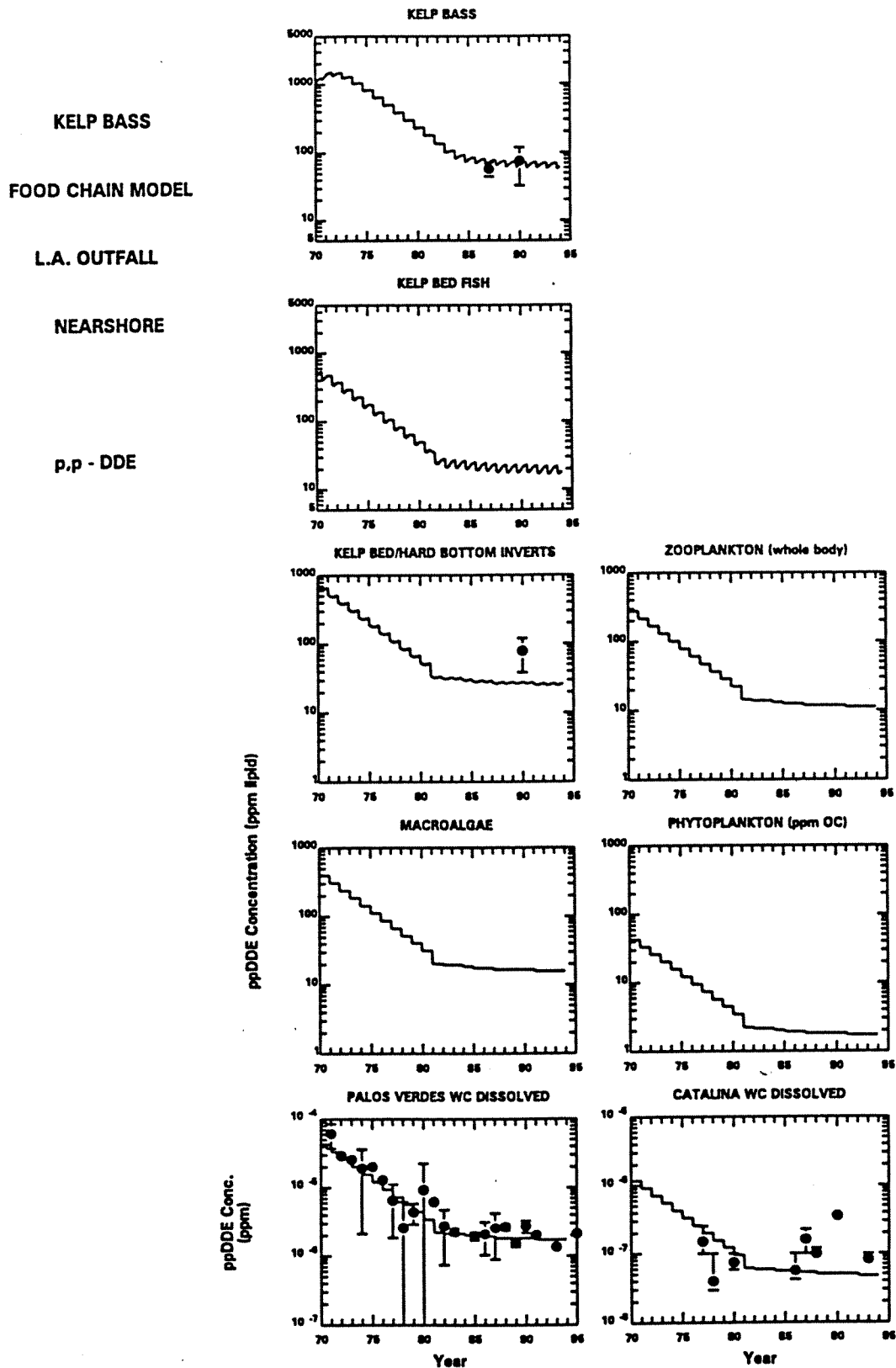
p,p - DDE
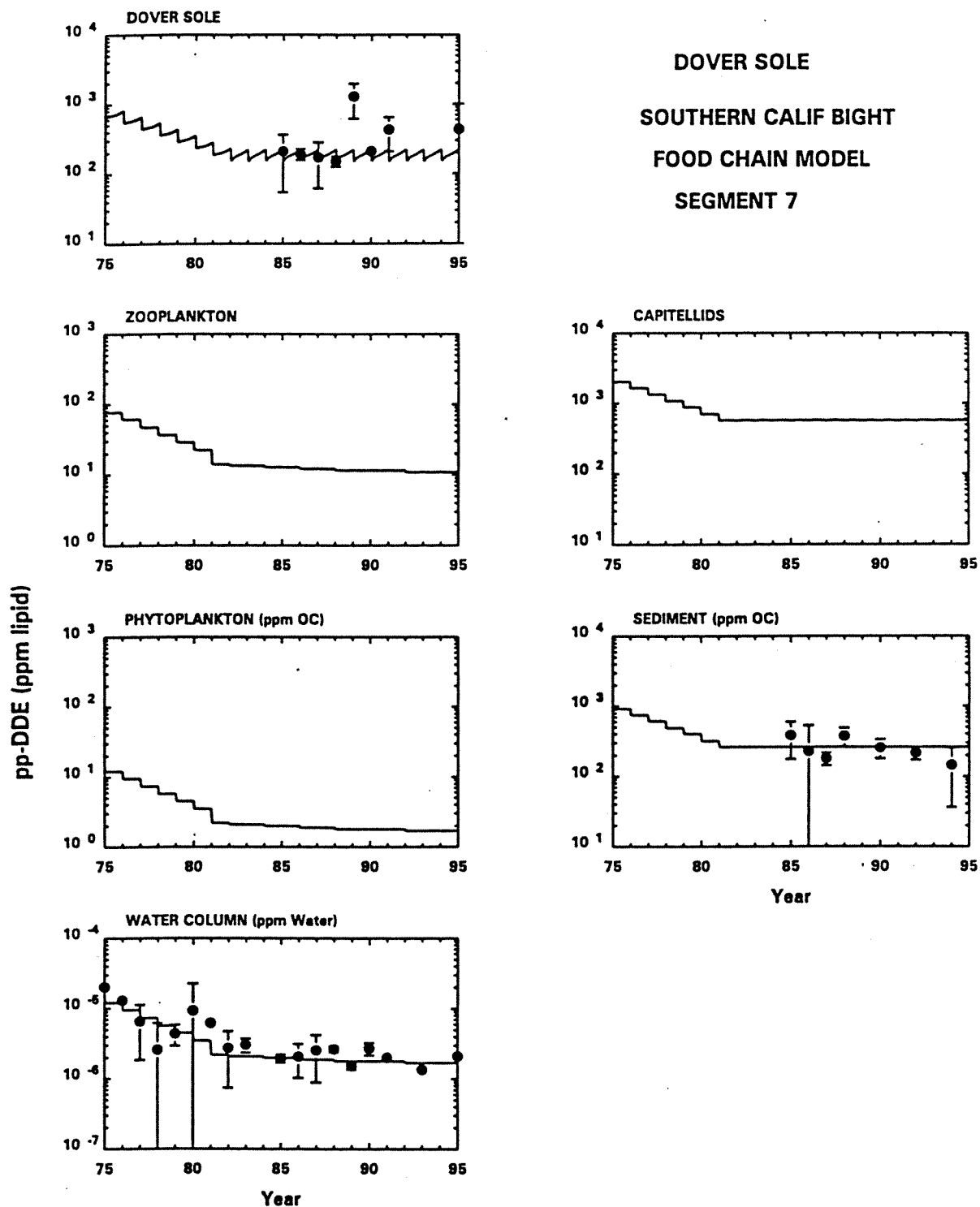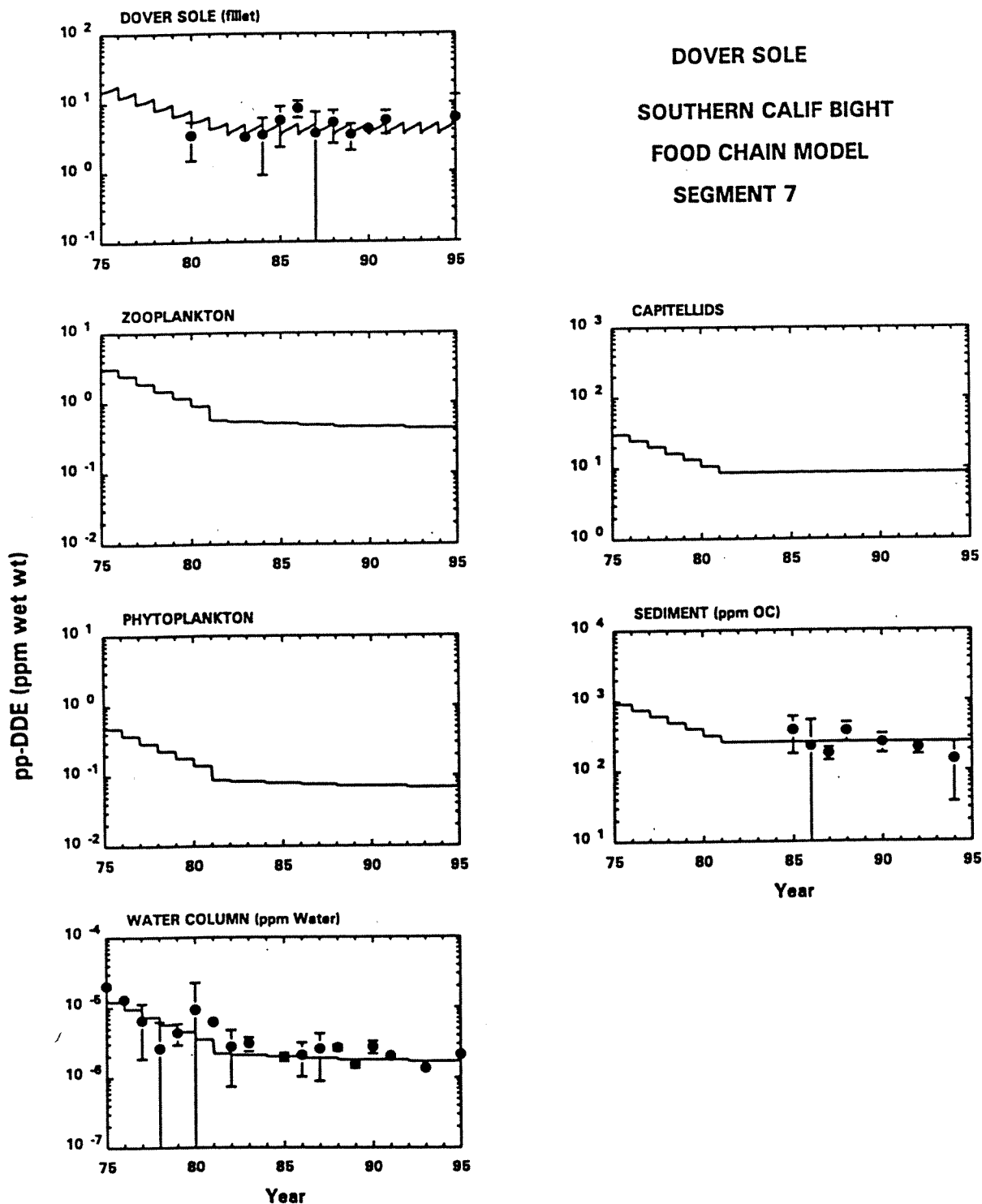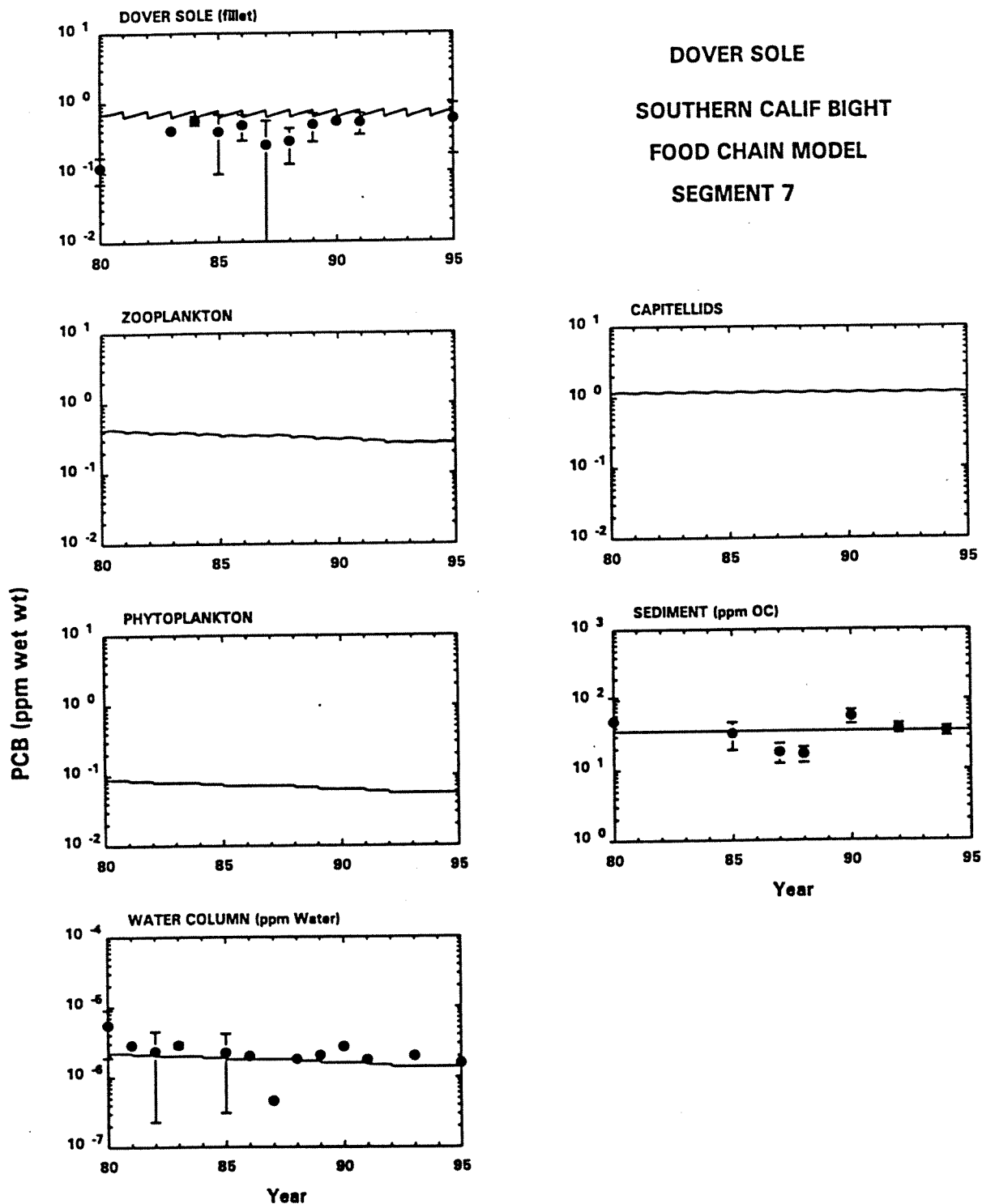

Figure 3-30. Computed lipid normalized p,p'DDE concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island). Data are arithmetic means +/- 2 standard errors of the mean.
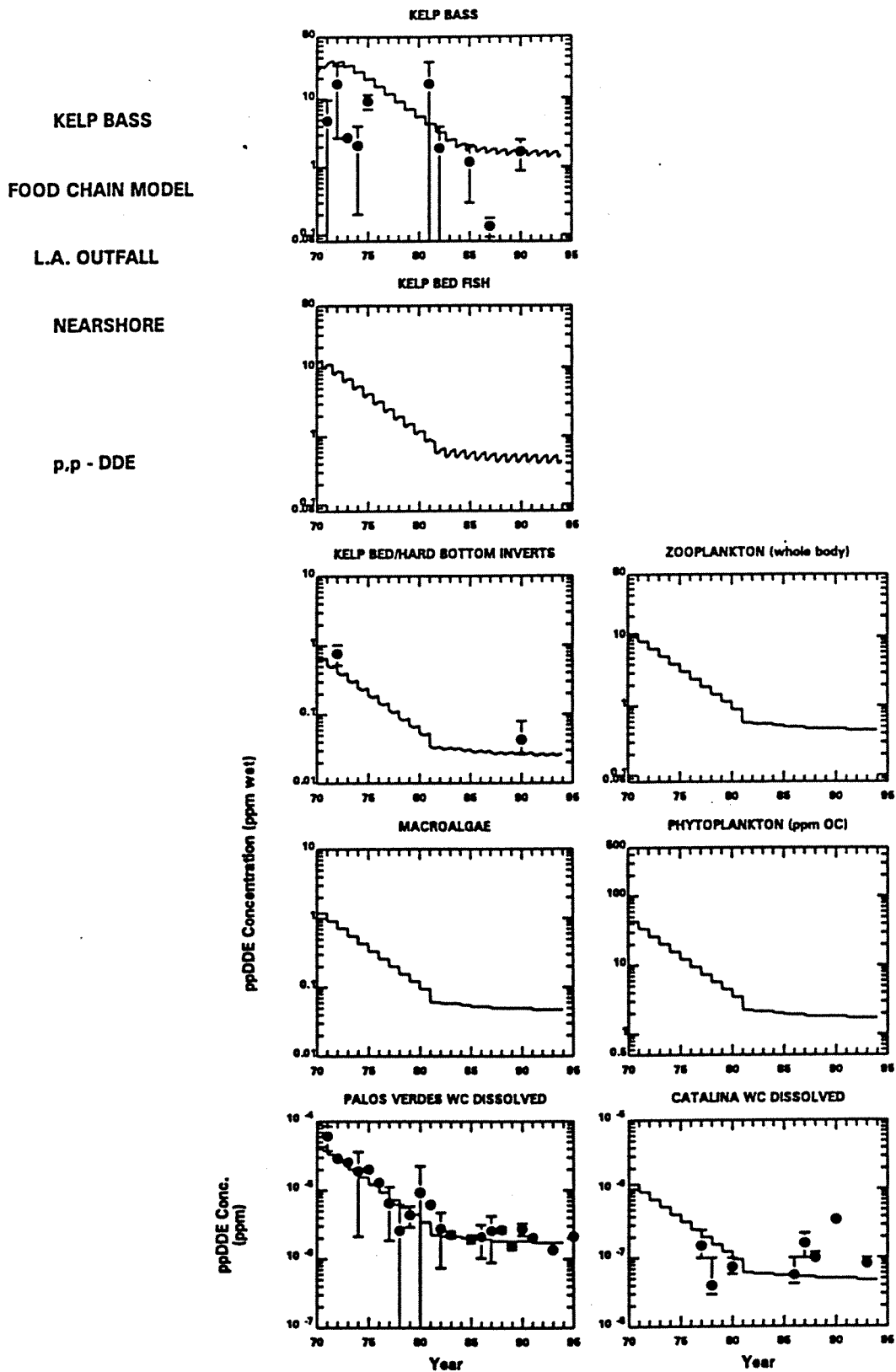
Figure 3-31. Computed wet weight p,p'DDE concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island). Data are arithmetic means +/- 2 standard errors of the mean.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE
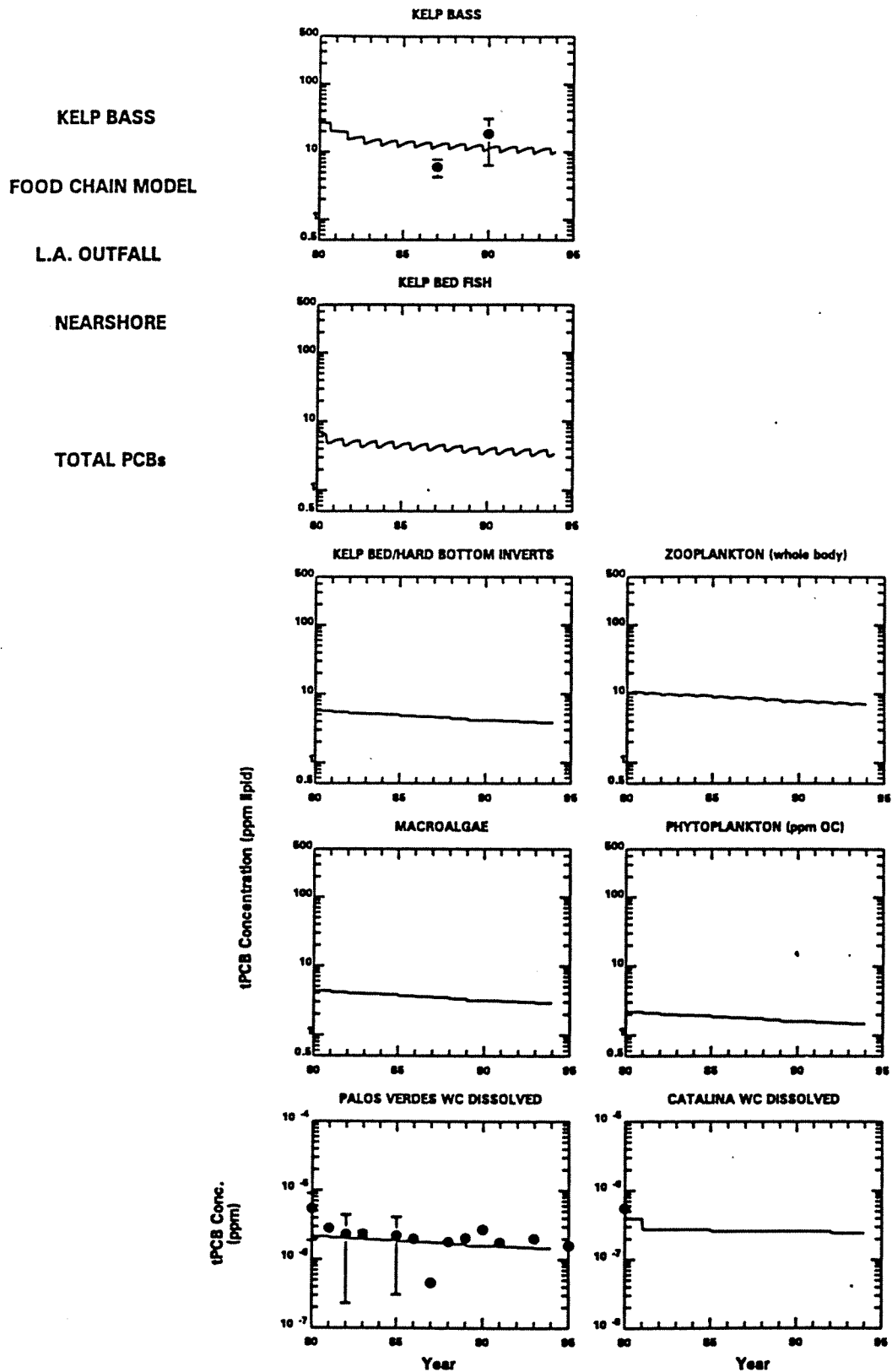
TOTAL PCBs

Figure 3-32. Computed lipid normalized total PCB concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island). Data are arithmetic means +/- 2 standard errors of the mean.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE
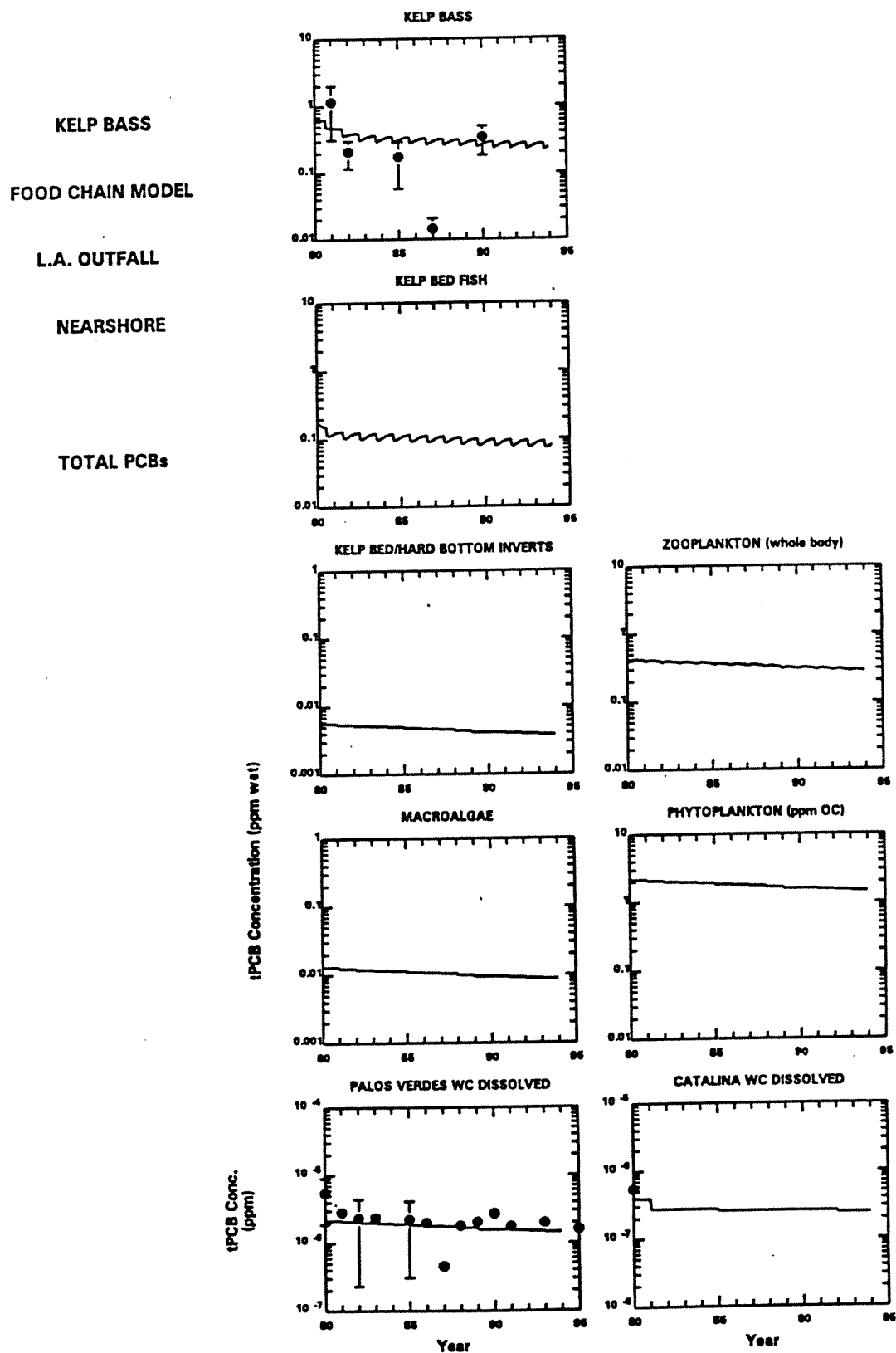
TOTAL PCBs

Figure 3-33. Computed wet weight total PCB concentrations (lines) and data (symbols) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island). Data are arithmetic means +/- 2 standard errors of the mean.
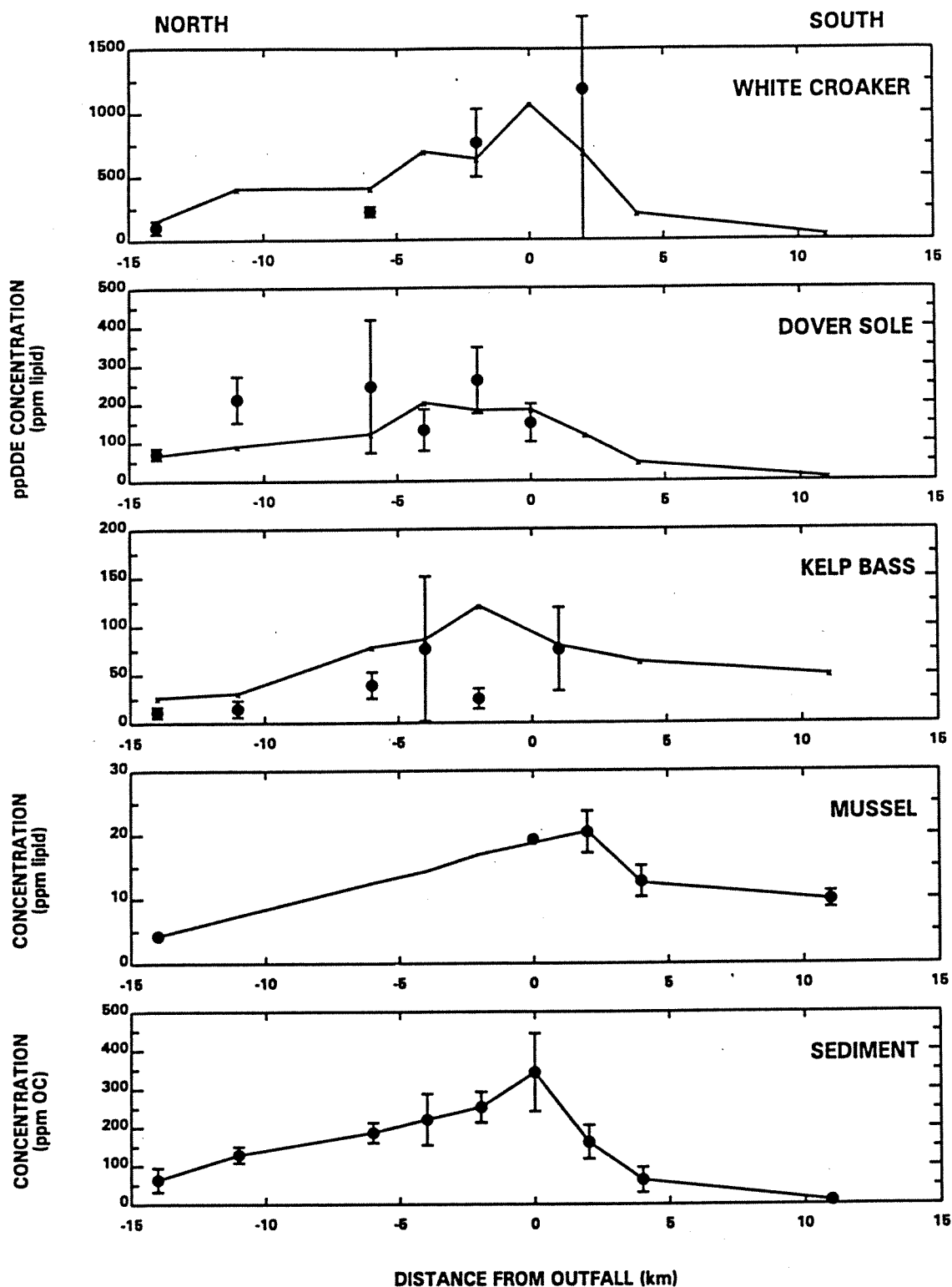
Figure 3-34. Comparison of computed and observed p,p'DDE concentrations for white croaker, Dover sole, kelp bass, mussels and sediments plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). Solid lines indicate steady-state p,p'DDE concentrations predicted by the food web model. Concentrations in fish fillets and mussels are expressed as ppm lipid and surficial sediment concentrations are expressed as ppm organic carbon. Data are arithmetic means +/- 2 standard errors of the mean.
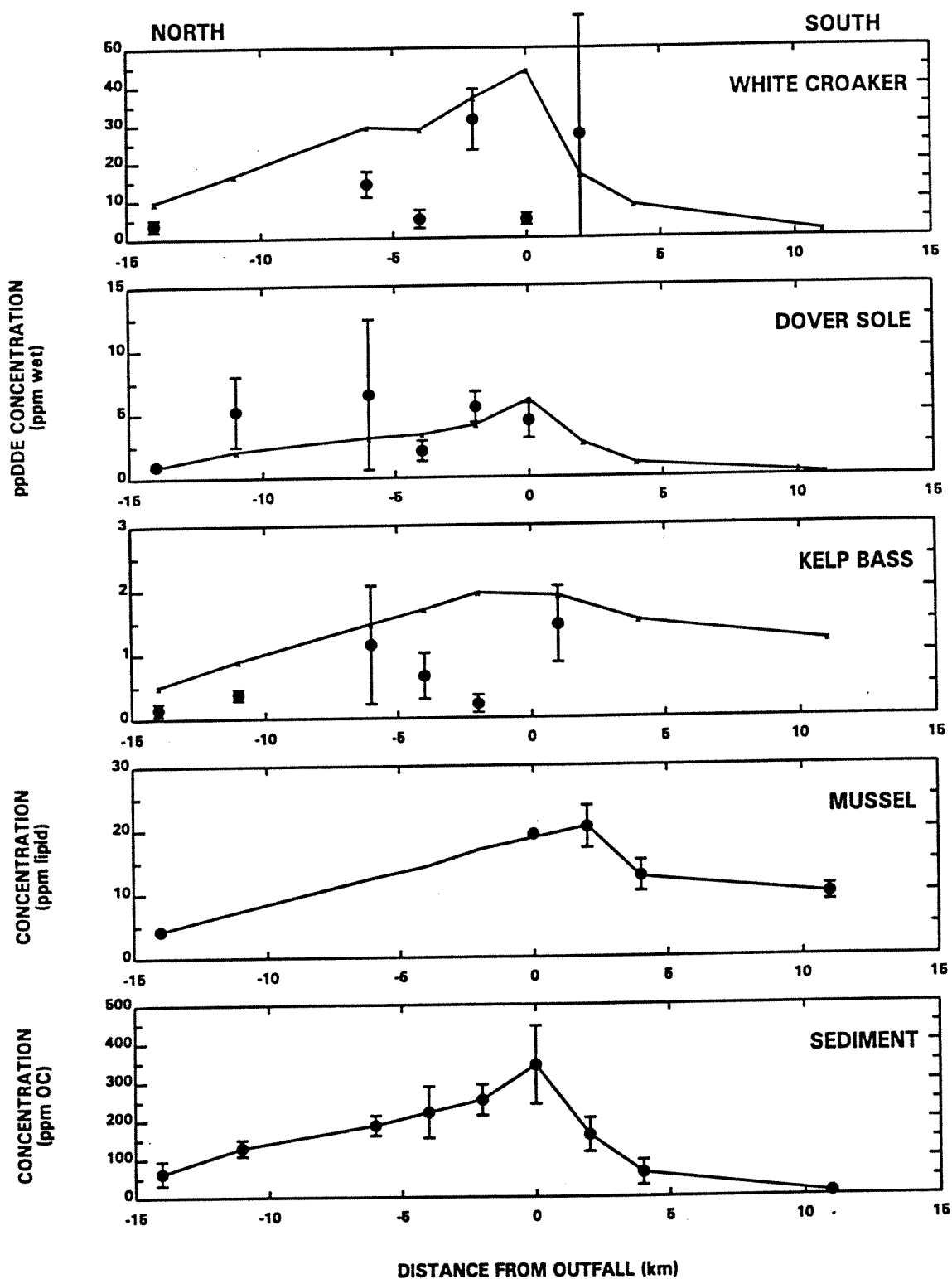
Figure 3-35. Comparison of computed and observed p,p'DDE concentrations for white croaker, Dover sole, kelp bass, mussels and sediments plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). Solid lines indicate steady-state p,p'DDE concentrations predicted by the food web model. Concentrations in fish fillets are expressed as ppm wet weight, mussels are expressed as ppm lipid and surficial sediment concentrations are expressed as ppm organic carbon. Data are arithmetic means +/- 2 standard errors of the mean.

comparison, the results of the white croaker and Dover sole model simulations presented above are indicated at -2 km in Figure 3-34; the kelp bass simulation presented above is indicated at +2 km in Figure 3-34.

Figure 3-34 includes all contaminant data for fillets with associated lipid contents. Figure 3-35 includes all wet weight-based contaminant data in fillets. Because lipid contents were not available for every contaminant measurement, some of the averages in Figure 3-35 include more data than the averages in Figure 3-34. The lipid contents used in the model were estimated using the lipid contents associated with the contaminant measurements in Figure 3-34. In some cases, lipid-based data are closer to the model results than wet weight-based data. This is because of the inclusion in Figure 3-35 of contaminant data with unknown lipid contents. Thus, Figure 3-34 provides the more appropriate model/data comparison. Figure 3-35 is included for comparison, but is subject to the uncertainty in lipid contents associated with the additional data in this figure.

The white croaker model computes p,p'DDE levels in the fish that are consistent with levels measured in the segment near the outfall (-2 km) and the segment 14 km to the north. Values computed at -6 km are within a factor of two of the data. The measured values at +2 km exhibit a great deal of variability, and the model results are within two standard errors of the data mean. Overall, comparison between model and data supports the assumptions and structure of the food web model, in particular the assumption that local populations of white croaker derive their contaminant loads from local sediments. On a wet-weight basis (Figure 3-35), results are similar, except for the presence of two additional data values at 0 and -4 km. These are much lower than the computed values. They were collected in 1985 by the Los Angeles County Sanitation District, and no lipid values are available for them. Thus, the appropriate lipid content for use in the model is not known. If these fish had relatively low lipid contents, then their wet weight-based concentration would be expected to be low. Given the uncertainty associated with their lipid contents, it is inappropriate to compare the model results with these data.

p,p'DDE levels measured in the Dover sole data exhibit less of a spatial gradient than the levels measured in the white croaker over the region from the outfall to +14 km. Consistent with this, the gradient computed by the Dover sole model is also weaker than the gradient computed for the white croaker. Computed levels in the Dover sole sometimes overestimate and sometimes underestimate measured levels but generally run through the range of the means of

the data. Thus, as for the white croaker, the model results support the assumptions and structure of the food web model, in particular the assumption that local populations of Dover sole derive their contaminant loads from local sediments.

The gradient in p,p'DDE levels measured in the kelp bass between the outfall and -14 km is steeper than for the Dover sole and shallower than for the white croaker. The model computes a gradient that is consistent with this. However, the model generally overestimates measured levels between -2 km and -14 km by up to a factor of two (lipid-based) and three (wet weight-based) (except for one point at -2 km). This could be due to the method by which the water column levels were estimated. Based on the mussel data presented in Figure 3-34, mussel levels were linearly extrapolated between mile +2 and -14. Alternatively, an exponential decline would have resulted in lower levels in the segments from 0 to -11 km, which would have improved the fit between model and data.

Thus, p,p'DDE levels in the local waters of the Palos Verdes shelf are sufficient to produce the p,p'DDE levels observed in the kelp bass caught on the shelf. The results of the simulations in which kelp bass were exposed to levels characteristic of an offshore site indicated that the exposure levels found near the Palos Verdes Shelf are necessary to achieve the levels observed in the kelp bass (Santa Catalina Island; see above).

3.9 SENSITIVITY AND UNCERTAINTY ANALYSES

Two additional studies of the potential variation in model results were performed: the sensitivity to two-fold variation in water column contaminant concentrations, and the uncertainty in the model calculations that flows from uncertainty in the model parameter estimates.

Sensitivity to variation in water column contaminant levels. As described in Section 2.5, water column concentrations were estimated using mussel concentrations and a bioaccumulation factor. There is some uncertainty in the concentrations of p,p'DDE and PCBs in the water column, due to limitations in the data available to determine bioaccumulation factors for mussels (see Section 2.5). Therefore, the models for all three fish species were each run using double and one-half the best estimate of water column concentrations, for both p,p'DDE and PCBs. Model results are presented for all three species and both chemicals in Figures 3-36 and 3-37.
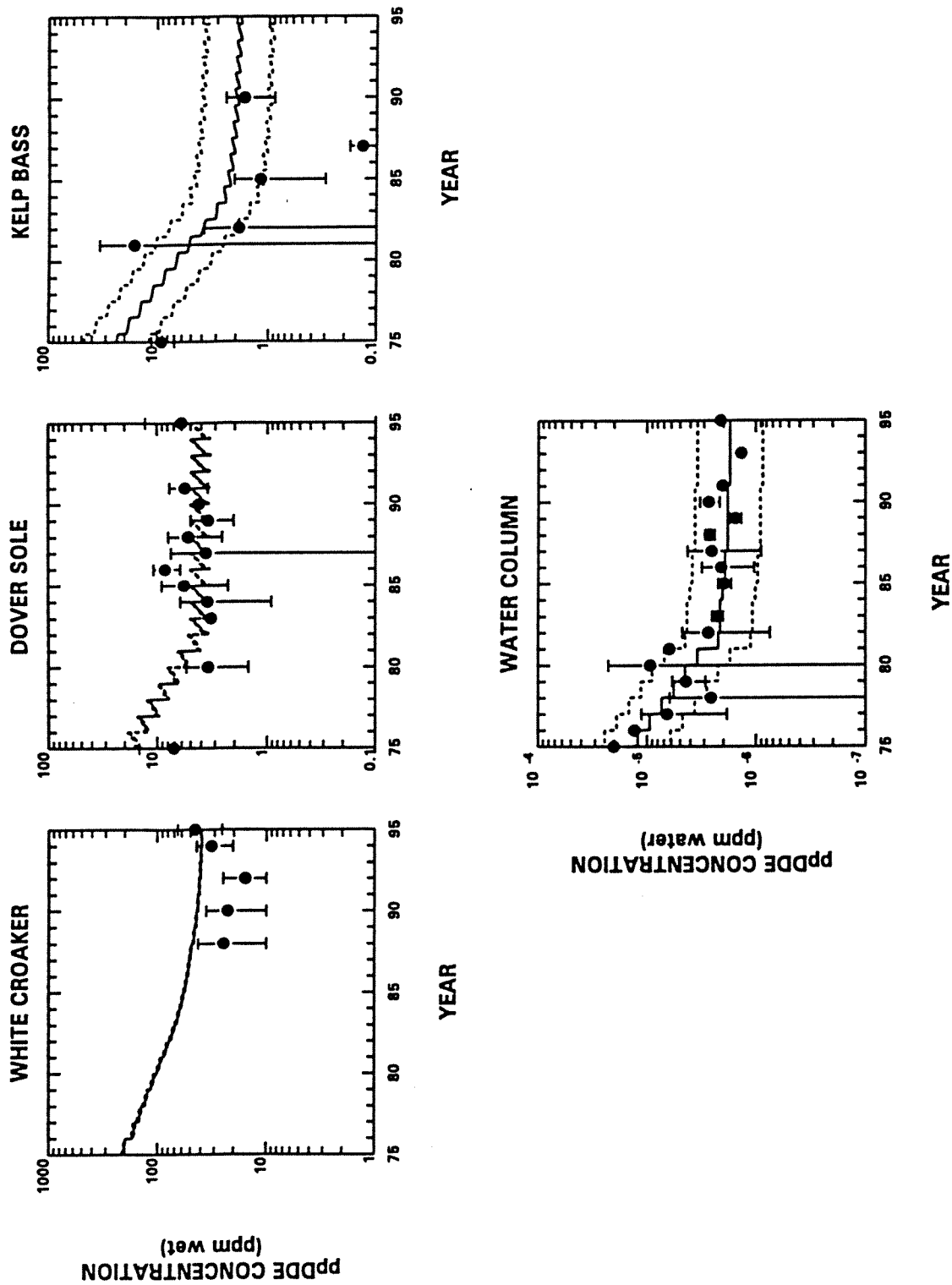
Figure 3-36. Computed wet weight p,p'DDE concentrations (lines) and data (symbols) for white croaker, Dover sole, kelp bass and water column for zone of high contamination. Solid lines indicate model results presented in Sections 3.5 through 3.7; dashed lines indicate computed concentrations when water column concentrations are doubled and halved (ppm wet weight; data are arithmetic means +/- 2 standard errors of the mean).
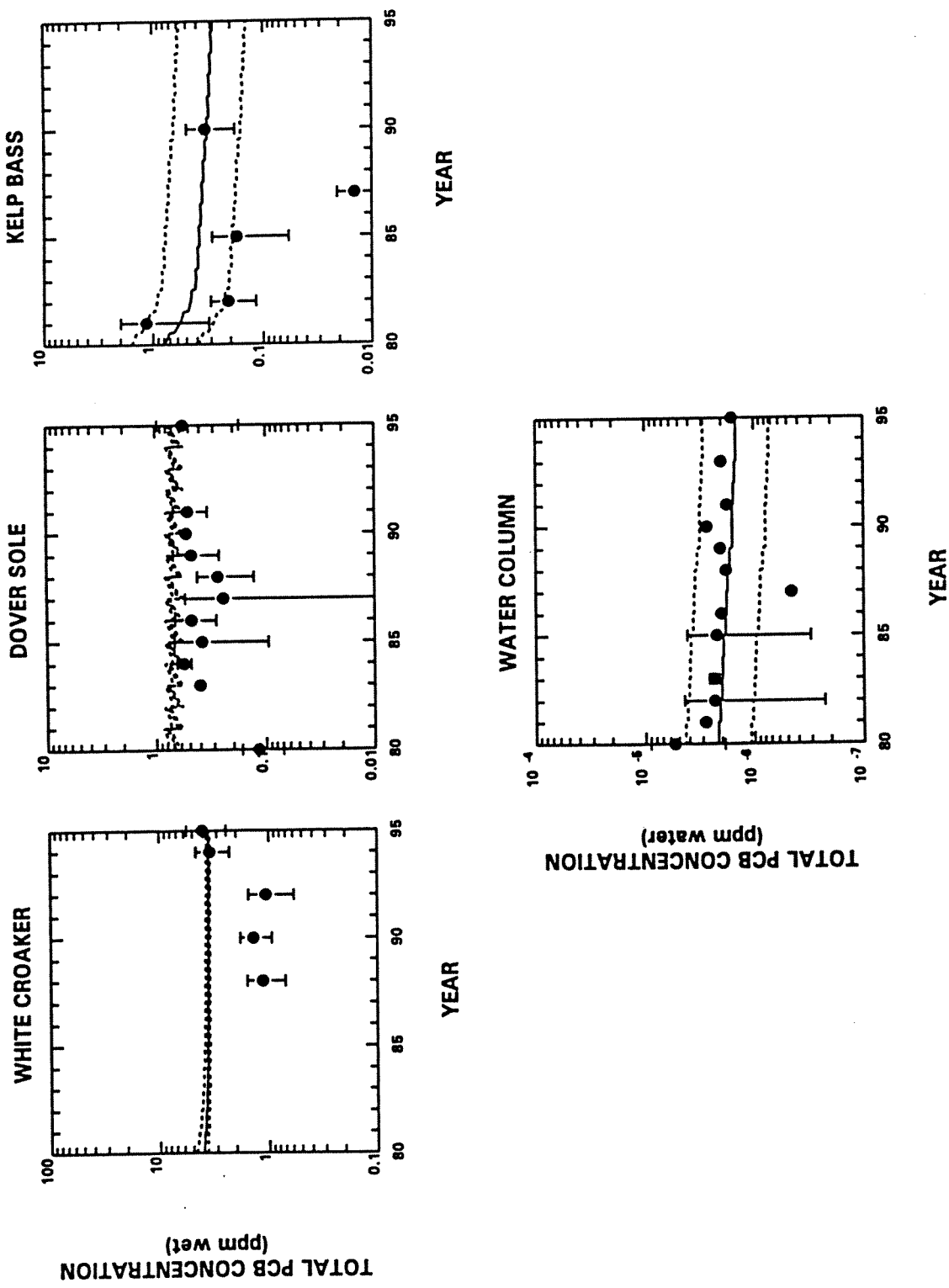
Figure 3-37. Computed wet weight Total PCB concentrations (lines) and data (symbols) for white croaker, Dover sole, kelp bass and water column for zone of high contamination. Solid lines indicate model results presented in Sections 3.5 through 3.7; dashed lines indicate computed concentrations when water column concentrations are doubled and halved (ppm wet weight; data are arithmetic means +/- 2 standard errors of the mean).

The model results for white croaker and Dover sole are seen to be insensitive to variation in water column concentrations, because most of their contaminant loads originate from the sediment. In contrast, the two-fold changes in water column concentration translate directly into two-fold variation in kelp bass levels. Even with this variation, model results are still within the limits of the data.

Uncertainty in model parameter values. A source of uncertainty in the model calculations is the uncertainty of the parameter values chosen for the bioenergetics and toxicokinetics components of the model. The effects of variation in these parameters on model results for p,p'DDE was studied using a Monte Carlo analysis.

First, a set of parameters was chosen for study. The choice was based on the expected importance of each parameter to model results, on the expected degree of variation in its value, and on the information available to define the distribution. The expected importance is based upon an understanding of the model structure. For example, the net growth efficiency is a key parameter in the model. The uncertainty in net growth efficiency can be studied by varying any of several growth or respiration-related parameters, and variation in any of these has a similar effect on the results; thus, only one respiration or growth parameter needs to be varied.

Second, the coefficient of variation (CV) for each of these parameters was estimated, based either on literature values or on data in the HydroQual database. The Monte Carlo analysis is designed to study uncertainty in the best estimate of each parameter, and therefore the standard error of the data was used to calculate the CV. A uniform distribution was used if no information was available concerning the shape of the distribution of a parameter. In addition, certain parameters were given absolute limits. For example, gill assimilation efficiencies less than 0.3 were not allowed. The parameters, their coefficients of variation and absolute limits, and the sources of the information on the CV's are indicated in Table 3-5.

Water column concentrations in the SCB were computed by dividing the measured mussel concentrations by the measured BAF values. The uncertainty associated with estimates of water column concentration is due to variation in both of these variables; only nine mussel measurements and six BAF measurements were available. To estimate the uncertainty in water column concentration due to the limited number of measurements, a resampling analysis was performed using the distribution of mussel concentrations and BAFs. Two hundred water column concentrations were calculated, each based on a sampling of 6 BAF values and 9 mussel

concentrations from their respective distributions. The coefficient of variation of the distribution of the 200 estimates of water concentration was used in the bioaccumulation Monte Carlo analysis. This value is given in Table 3-5.

TABLE 3-5. Parameter Uncertainty Considered in the Analysis of Model Uncertainty for p,p'DDE

Parameter	Species	Distribution	Coefficient of variation	Absolute Limits	Source
Net growth efficiency	All fish	normal	0.10	-	Humphreys 1979
Fraction lipid	White croaker	log normal	0.315	0.16-0.34	Species-specific information in HydroQual database
Fraction lipid	Kelp bass	log normal	0.417	0.005-0.37	
Fraction lipid	Dover sole	log normal	0.207	0.004-0.11	Species-specific information in HydroQual database
Log K_{ow}	All	normal	0.0016	-	de Bruijn et al. 1988
Feeding preference: day of the second year of life when the switch from zooplankton to benthic invertebrates occurs	Dover sole	uniform	-	1 to 366	This allows the switch to occur any time during the second year of life, following Horton 1989, Hagerman 1952, Hunter et al. 1990
Assimilation efficiency - gill	All fish	uniform	-	0.3 - 1.0	see Section 3.4.2
Biota/Sediment Accumulation Factor	Benthic invertebrates	uniform	-	0.87 - 4.6	see Section 3.3
Water column concentrations	All fish	log normal	0.45	-	see text

The model was run 200 times. For each run, a value for each of the chosen parameters was selected from the distribution appropriate for that parameter (normal distributions were used for each parameter). All model results were stored, and then plotted with the data.

The results of the Monte Carlo analysis are given in Table 3-6. To characterize the uncertainty associated with the estimates of the measured and computed mean fish concentrations, the 90 percent confidence limits on the mean of the data (computed using Land's method; Gilbert 1975) were compared with the range containing 90 percent of the Monte Carlo model results. The shapes of the data distributions were based on the data themselves; log normal for white croaker and kelp bass, normal for Dover sole.

There is extensive overlap between the 90 percent ranges of model and data for all three species (Table 3-6). For all three species, the high end of the 90 percent range is greater than the high end of the data, suggesting that for some possible sets of parameter values, the model overestimates measured levels in fish. Only in the case of the Dover sole does the lower end of the 90 percent range extent below the lower end of the data range, and by less than a factor of two. Thus, it appears unlikely that the model computes lower contaminant levels than are observed in the fish, and therefore it is unlikely that the fish are exposed to sources of contaminants other than those available in local sediments and water.

Table 3-6. Results of the Monte Carlo Analysis of p,p'DDE Concentrations in Fish

	Data 90% confidence interval	Model range containing 90% of simulated concentrations
White croaker	17 - 45	19 - 93
Dover sole	4.4 - 6.5	2.5 - 10.6
Kelp bass	1.0 - 2.5	1.4 - 5.8

Thus, while parameter uncertainty results in increased uncertainty in model calculations, the existence of that uncertainty does not effect the strength of the conclusions drawn from the data.

SECTION 4

FOOD CHAIN MODELS OF SEA LIONS

4.1 MODEL STRUCTURE

The dynamics of contaminant accumulation and loss in mammals differ from those of fish and aquatic invertebrates. The differences result from basic differences in physiology and behavior. Because hydrophobic organic contaminants more readily transfer across the gill surface than across the lung surface, the fish gills provide a potentially important mechanism of contaminant elimination whereas elimination via the lungs of a mammal is insignificant. The nursing of young provides a means of contaminant cycling in mammals that transfers a large fraction of adult contaminant body burden to the young. No analogous mechanism exists in fish and aquatic invertebrates.

The female sea lion is considered to have three body components among which contaminant partitions: lipid, milk and non-lipid body tissue (referred to as the aqueous component). p,p'DDE or PCBs taken up from food (or water) is distributed among the components in a proportion defined by a lipid:aqueous equilibrium partition coefficient and the lipid content of the milk. In addition, contaminant is distributed into a fetus during the time period of pregnancy. The rate at which contaminant is taken up from food is defined by a feeding rate, a prey contaminant concentration and the contaminant assimilation efficiency. The transfer of contaminant from mother to nursing young is defined by a nursing rate and the contaminant concentration in the milk. Excretion of the contaminant is defined by a rate constant and the contaminant concentration in the aqueous component. As with the models for fish and birds, the feeding rate is determined from a simple energy balance; the rate at which an animal uses energy for growth and metabolism is equal to the product of an efficiency of energy uptake and the rate of energy intake. Growth, metabolism and the efficiency of energy intake are measurable quantities for which data exist. Values determined from field and lab studies are used to compute the rate of energy intake. An energy density of prey animals (kcal/g) is used to convert energy intake rate (kcal/d) to a feeding rate (g/d). The equations that comprise the model are presented in Appendix 1.

Adult male sea lions are not considered by the model for two reasons: 1) these animals were excluded from the major studies of DDT and PCB contamination in Channel Islands sea
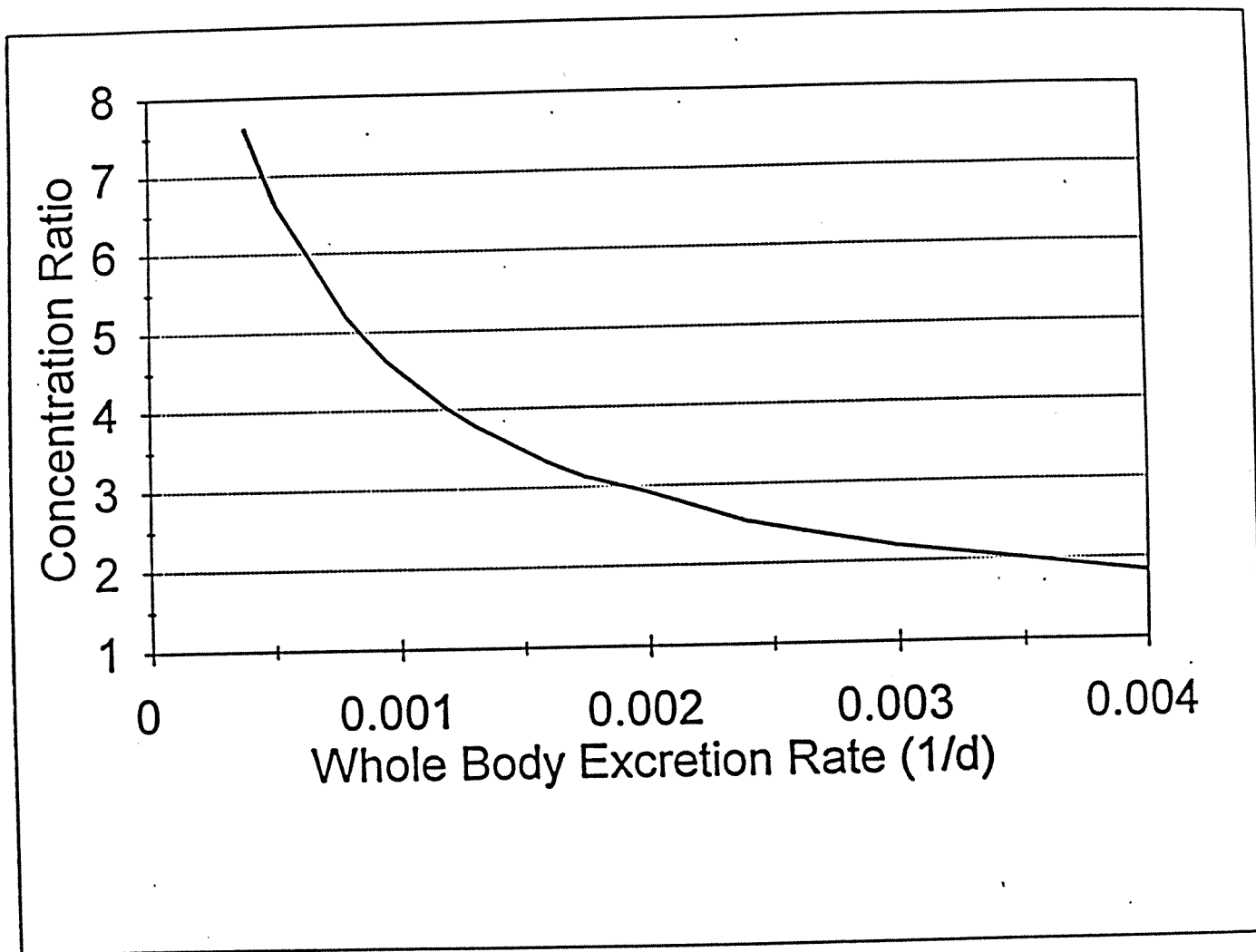
Figures 4-1. Ratio of the averages of p,p'DDE or PCB concentration in 4 to 7 and 9 to 12 year old female sea lions in relation to the whole body excretion rate. Note that excretion rates are those of a 100 Kg animal. Rates for smaller animals are proportional to these values.

lions; and 2) the annual migration of these animals between breeding sites in the Channel Islands and the Northwest US coast complicates estimation of their dietary contaminant concentration.

4.2 ESTIMATION OF TOXICOKINETIC PARAMETERS

4.2.1 Gut Transfer

The transfer of contaminant across the gut wall is modeled as described in Section 3.4.1. The relationship between contaminant and food energy assimilation is likely to apply to all animals because the mechanisms of uptake are generalizable across species (Standaert 1988). Accordingly, as in all the other models, we have assumed that the contaminant:food energy assimilation efficiency ratio is equal to 0.75 for p,p'DDE and 1 for PCBs.

4.2.2 Internal Partitioning

The equilibrium partitioning of hydrophobic contaminants between lipid and aqueous phases is typically described as being equal to K_{ow}. Comparisons of fish lipid:water partition coefficients (or, equivalently, bioconcentration factors) and K_{ow} (e.g., Chiou 1985) form the basis for this assumption. Although studies conducted with high K_{ow} compounds suggest that K_{ow} begins to overestimate the lipid:water partition coefficient as log K_{ow} increases beyond about 6 (Gobas *et al.* 1988), the equality assumption has generally been extended to log K_{ow} values of 7, or greater. Any overestimation has little effect on computed bioaccumulation because it serves only to reduce the rate of excretion below a value that typically is already unimportant relative to growth dilution.

More important to the modeling is partitioning of the contaminant between the body lipid and the milk lipid. A compilation of published measurements (Table 4-1) shows little evidence of preferential partitioning. Ratios of the concentrations of p,p'DDE and PCBs in milk and body lipid are approximately equal to 1. On the basis of this information we have used p,p'DDE and PCBs fat:aqueous partition coefficients for body fat and milk that are equal to K_{ow}.
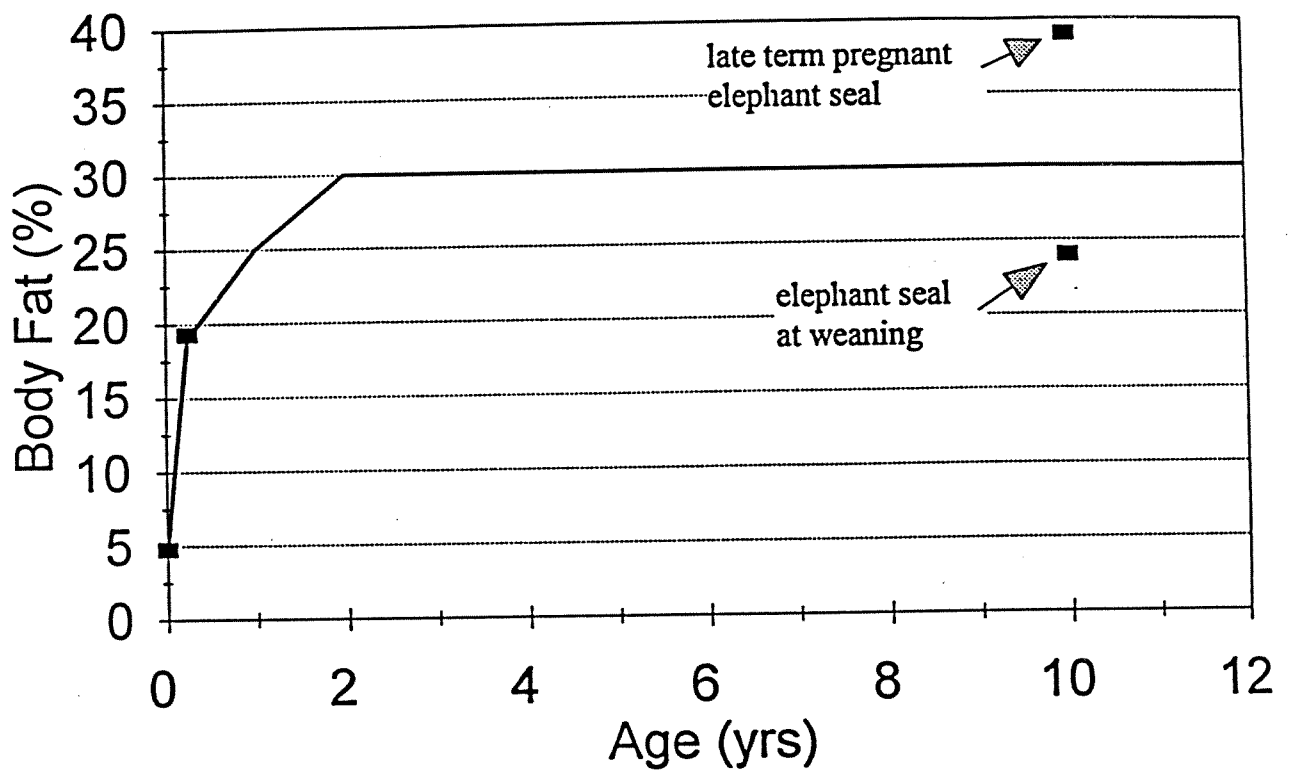
California Sea Lion Data: Oftedal et al., 1987.
Elephant Seal Data: Costa et al., 1986.

Figures 4-3. Observed (symbols) and estimated (line) whole-body fat content of female
sea lions as a function of age.

Table 4-2. Measurements of p,p'DDE and PCBs Half-Lives in Various Mammals

Chemical	Species	Sex	Age (yrs.)	Weight (g)	Fraction Lipid	Growth Corrected (Yes/No)	Half-Life (days)	Reference	Notes
PCBs	mink	M	>10	NS	NS	N	100	1	rate calculated from adipose tissue data. Chronic feeding study with contaminated fish.
p,p'DDE	rat	NS	NS	320 to 390	0.1	N	48	2	rate calculated from adipose tissue data. Single iv dose study.
p,p'DDE	rat	NS	NS	220 to 230	NS	Y	120	3	whole body loss rate following single iv dose.
PCBs	humans	NS	17, 25, 33	NS	NS	N	1160 to 1680	4	exposed in Yu-Cheng rice oil poisoning. Data also for individual congeners.

NS = not specified
1. Hornshaw et al., 1983
2. Mühlebach et al., 1985.
3. Mühlebach et al. 1991.
4. Ryan et al., 1993.

Table 4-1. Ratios of Contaminant (DDTs or PCBs) Concentrations in the Lipid of Milk, Serum and Adipose Tissue (fat)

Animal	Chemical	Milk/Fat	Milk/Serum	Serum/Fat	Reference
Humans	PCBs		0.2		Jacobson *et al.* 1984
Humans	*p,p*'DDT	0.9	2.6	0.3	Kanja *et al.* 1992
	o,p'DDT	1.1	0.9	1.2	
	p,p'DDE	0.7	1.5	0.5	
Humans	PCBs		0.6		Mes *et al.* 1984
	HCB		0.8		
	p,p'DDE		0.8		
	p,p'DDT		1.5		
	dieldrin		0.3		
Humans	p,p'DDE			0.4	Mussalo-Rauhamaa 1991
	DDTs			0.3	
	PCBs			0.5	
	HCB			2	
Rhesus monkeys	PCB Aroclor 1016	1	0.6	1.7	Barsotti & Van Miller 1984
Range	p,p'DDE	0.7	0.8-1.5	0.4-0.5	
Range	DDT	0.9-1.1	0.9-2.6	0.3-1.2	
Range	PCBs	1	0.2-0.6	0.5-1.7	

4.2.3 Excretion

Few data are available to establish the excretion rate of p,p'DDE and PCBs by mammals. A summary of the data we were able to compile is shown in Table 4-2. p,p'DDE half-lives of 48 and 120 days were measured in laboratory rats. A PCB half-life of 100 days was measured for an adult mink. A study of adult humans accidentally exposed to PCBs indicated half-lives ranging from 1160 to 1680 days. Viewed together the PCB and p,p'DDE data indicate an increase in half-life with species size, a trend that is expected because of metabolic rate size scaling. Because the studies did not report the data needed to correct the half-lives for concentration reduction due to growth dilution and differences in whole-body fat content, the measurements only provide rough guidance for estimating a sea lion excretion rate. The model indicates that only juvenile sea lions (i.e., animals that have not experienced a lactation cycle) are sensitive to the excretion rate value. Milk production is the dominant contaminant loss mechanism in adult females. The difference in contaminant concentration between non-lactating and lactating animals is partially controlled by the excretion rate value. Therefore, the excretion
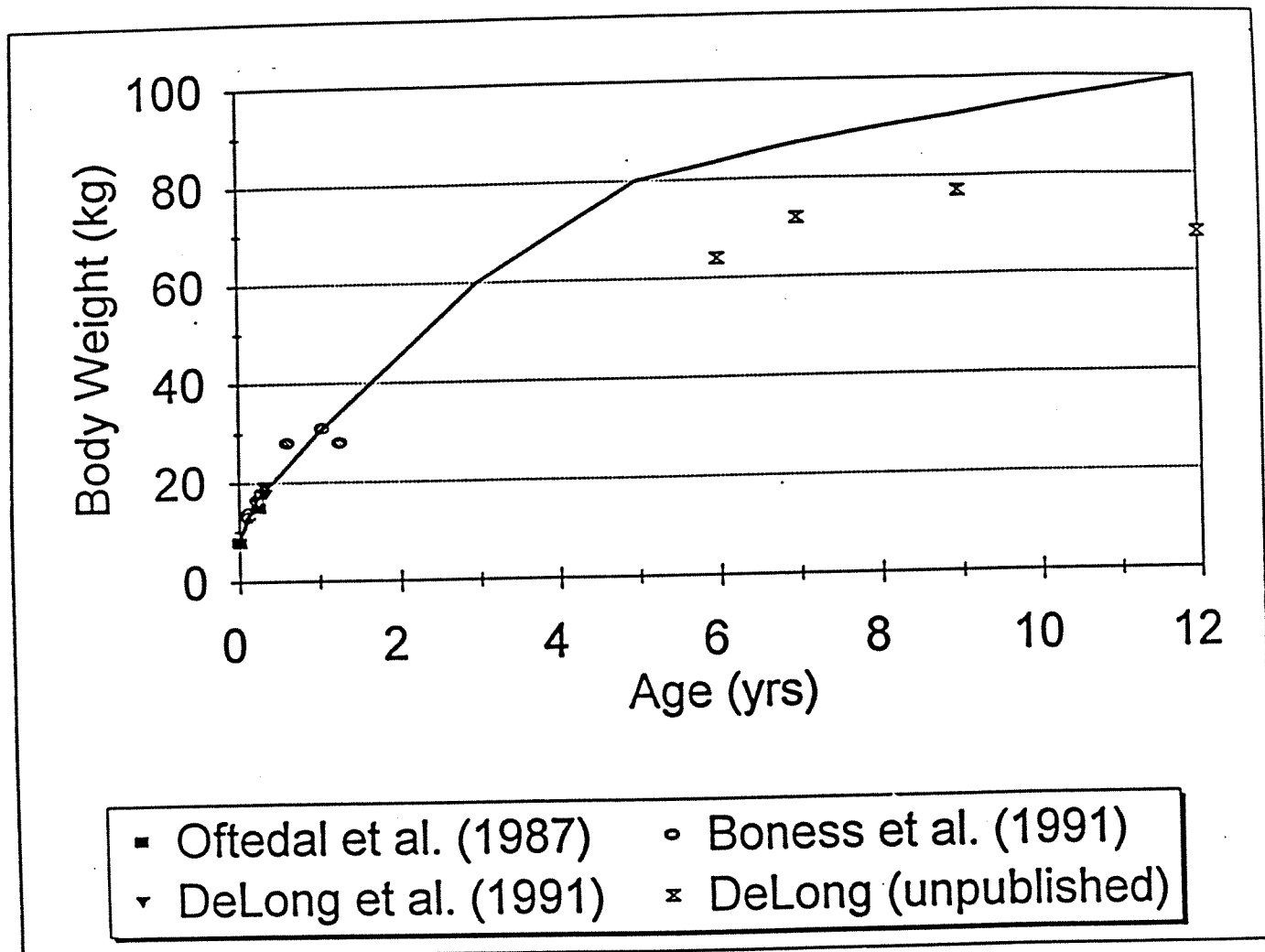
Figures 4-2. Observed (symbols) and estimated (line) weights of female California sea lions as a function of age.

rate may be estimated from the observed contaminant concentration differences between these groups. To illustrate this point the model was run with various excretion rate values and the steady-state concentrations in 4 to 7 year old animals were averaged and divided by the average of concentrations in 9 to 12 year old animals. Figure 4-1 presents this ratio in relation to the excretion rate of full grown (100kg) animals. Note that the line shown is representative of both PCBs and p,p'DDE. This is so because the losses through milk production are essentially the same for both chemicals, a consequence of lipid-aqueous phase partition coefficients high enough to keep essentially all of the chemical in the lipid phase. Note also that the excretion rates for younger animals are proportional to those shown, depending on differences in metabolic rate and whole-body fat content (see equations in Appendix 1). Figure 4-1 and the measured contaminant concentrations were used to establish excretion rates. The rates for 100 kg animals were 4.5×10^{-4} d^{-1} for p,p'DDE and 1.5×10^{-3} for PCBs. These are equivalent to half-lives of about 1,600 days for p,p'DDE and 500 days for PCBs.

4.3   ESTIMATION OF PHYSIOLOGICAL PARAMETERS

4.3.1   Growth

Full-grown female California sea lions weigh about 100 kg. At birth the female weighs about 8 kg (Oftedal et al. 1987b). During the first year of life the sea lion grows rapidly, reaching a weight of about 30 kg (Figure 4-2). The rate of growth from this age to adulthood is uncertain due to a lack of data. Measurements of five individuals ages 6 to 12 have been taken by Robert DeLong (National Marine Fisheries Service, unpublished data) and are shown on Figure 4-2. The weights of these animals are probably low because they were in captivity for several days and were somewhat dehydrated. The line shown on Figure 4-2 indicates the weight-age relationship used in the model.

The whole-body fat content of female sea lions increases from about 5 percent at birth to about 20 percent at age 3 months (Oftedal et al. 1987b). The fat content of older animals probably varies seasonally in response to temperature change, food availability, pregnancy and lactation. Data for adult elephant seals (10 years old) who, unlike the sea lion, do not feed during a several week nursing period indicate fat contents varying between about 38 percent in late term pregnancy and 24 percent at weaning (Costa et al. 1986). In the absence of data on whole-body fat content of juvenile and adult sea lions we have assumed an average value of 30 percent that is reached at two years of age (Figure 4-3).

4.3.2 Metabolism

The resting metabolic rate of animals is related to their size or weight. Allometric functions using weight as the dependent variable have been developed from data for numerous species. The generally accepted relationship for homeotherms was developed by Kleiber (1961). This relationship predicts values that agree closely with experimental measurements for sea lions. For example, a study of the resting metabolic rate of 6 sea lions ranging in weight from 26 to 38 kg yielded an average rate of 0.15 kilojoules per gram wet weight per day (kJ/g-d) (Butler *et al.* 1992). Using the average weight of these animals (32 kg), Klieber's equation predicts this value exactly. An earlier study with a female about the same weight (31.4 kg) reported a value of 0.22 kJ/g-d (Luecke *et al.* 1975).

A 1983 to 1984 study of the field metabolic rate of female California sea lions (Costa *et al.* 1991) found that the at-sea metabolic rate was proportional to Kleiber's equation. In the El Niño year of 1983 the at-sea rate was 6.9 times the predicted resting rate. The following year this rate was 4.8 times the basal rate. The authors indicate that the difference between years was due to a greater than normal energy expenditure in foraging during the 1983 El Niño. They also note that the 1984 value may be slightly above a typical value because of residual impacts of El Niño. In the model we have assumed an at-sea metabolic rate four times the resting rate. Consistent with Costa and coworkers we have assumed an on-shore metabolic rate of two times the resting rate. Because the females spend about equal times on-shore and off-shore we used an average metabolic rate that is three times the resting rate. The equation relating metabolic rate (R) in units kJ/g-d to body weight (W) in g is as follows.

$$R = 6W^{-0.249}$$

4.3.3 Pup Production

Female sea lions reach sexual maturity at age four to five with first parturition occurring at age five to six (Riedman 1990). Because of a high rate of premature pupping in younger sexually-mature females, the average age of first full-term parturition is likely to be somewhat higher. Birth of a single pup occurs sometime between mid May and late June (Odell 1975). For the purposes of the model we have assumed that parturition occurs annually on June 15, beginning at age eight. Although the choice of an age at first full-term parturition is somewhat

arbitrary, the only important effect is to shift the age at which contaminant concentrations drop due to loss through lactation.

4.3.4 Lactation

The transfer of p,p'DDE and PCBs from mother to young is dependent on the rate of milk production and the fat content of the milk. The equation describing this transfer is presented in Appendix 1. Measurements of milk production rate (Oftedal *et al.* 1987a) indicate that the rate is approximately constant for the first two months post partum at about 0.7 kg/d. It increases to about 1 kg/d three to four months post partum. It probably remains at about this rate until the pups begin to feed at about six to seven months post partum. After this it declines. Although differences exist between individuals, the average period of lactation is about 11 months (Oftedal *et al.* 1987a). From this information the milk production-days post partum relationship shown on Figure 4-4 was developed and used in the model.

The composition of sea lion milk has been measured a few days, two months and three to four months post partum (Oftedal *et al.* 1987b) (Figure 4-5). The composition was essentially constant in the first two months. The major components of the milk were water (59 percent), fat (32 percent) and protein (9 percent). Consistent with data for phocids, the fat content of the milk increased to about 44 percent in the middle of the lactation period (3 to 4 months post partum). It most likely remains at this level for the remainder of the period of lactation (Oftedal *et al.* 1987a). We have used this information to define a relationship between milk fat and days post partum relationship that is used to establish the partitioning of p,p'DDE and PCBs between milk and body fat of the mother (Figure 4-6).

4.4   FEEDING HABITS

The diet of the California sea lion is comprised of fish and invertebrates. The sea lion opportunistically feeds on a diversity of species. Composition varies seasonally and annually based on prey availability and abundance.

Sea lion food habits have been studied at three islands off the Southern California coast. Studies of sea lions on San Clemente Island (Lowry *et al.* 1990), San Nicolas Island (Lowry *et al.* 1991), and San Miguel Island (Antonelis *et al.* 1984, DeLong *et al.* unpublished. 1993) analyzed scat (i.e. fecal) material to determine prey species. The predominant prey species were found to be northern anchovy, Pacific whiting, jack mackerel, rockfish, market squid, red crab,
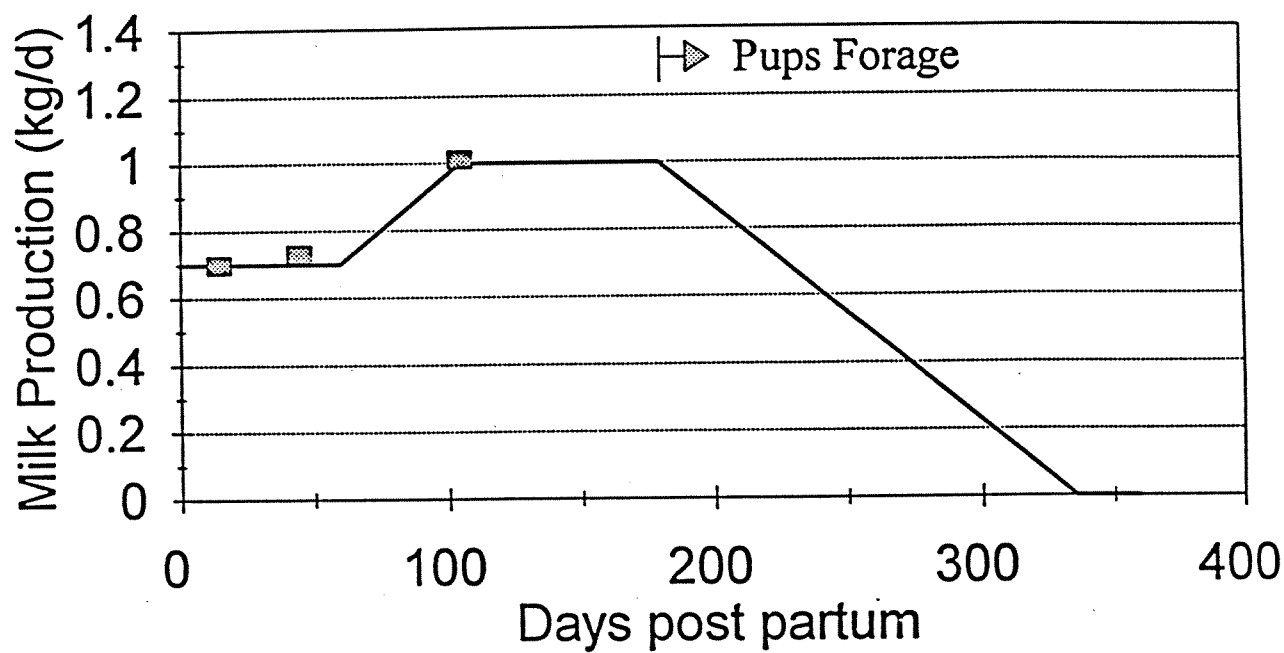
Figure 4-4. Sea lion milk production-days post partum relationship used in model.

carbohydrate (0.70%)
protein (8.60%)

0 to 60 days
post partum

fat (31.70%)

water (59.00%)

carbohydrate (0.90%)
protein (9.60%)

90 to 120 days
post partum

water (45.80%)

fat (43.70%)
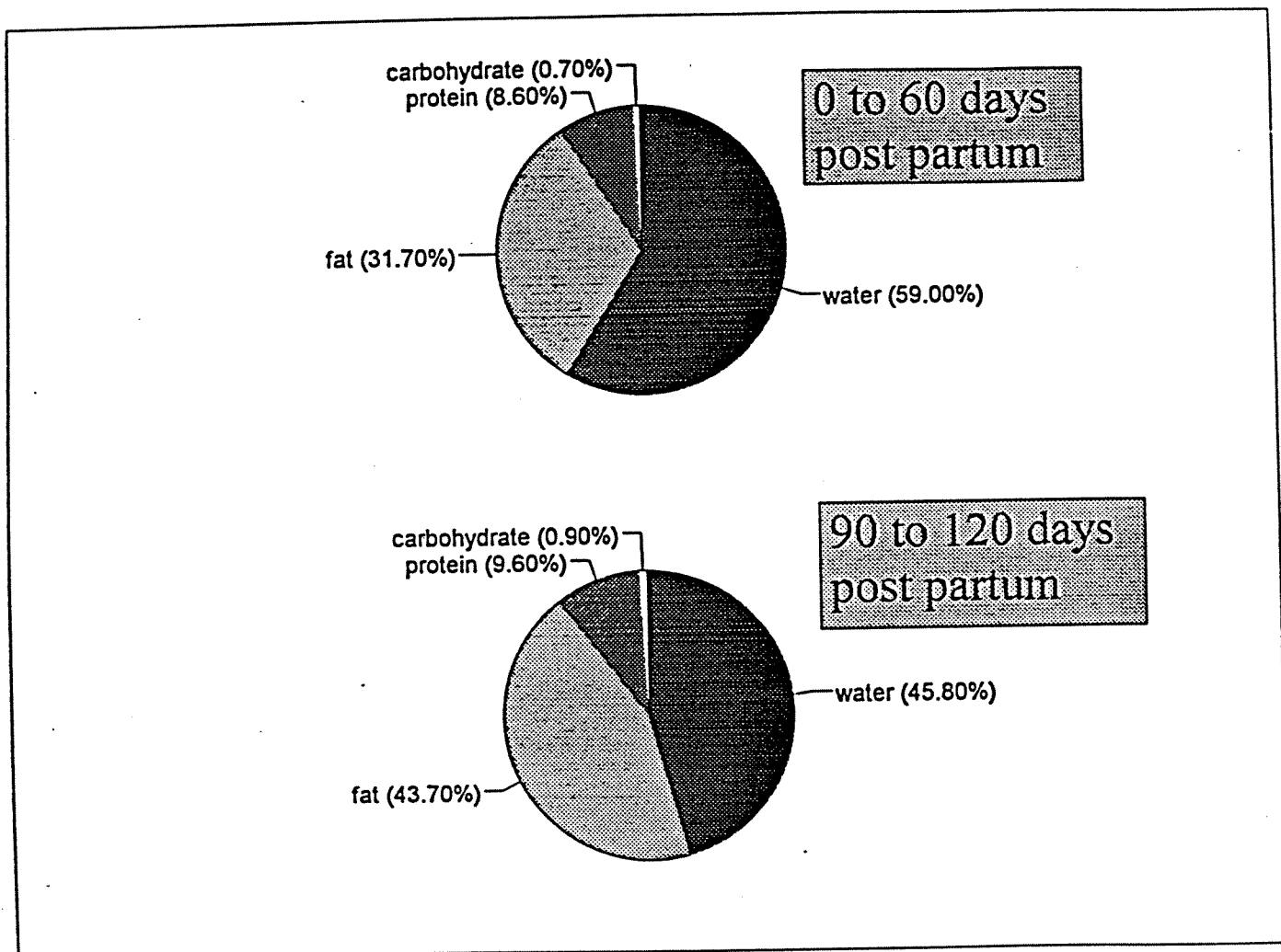
Figure 4-5. The composition of sea lion milk post partum.
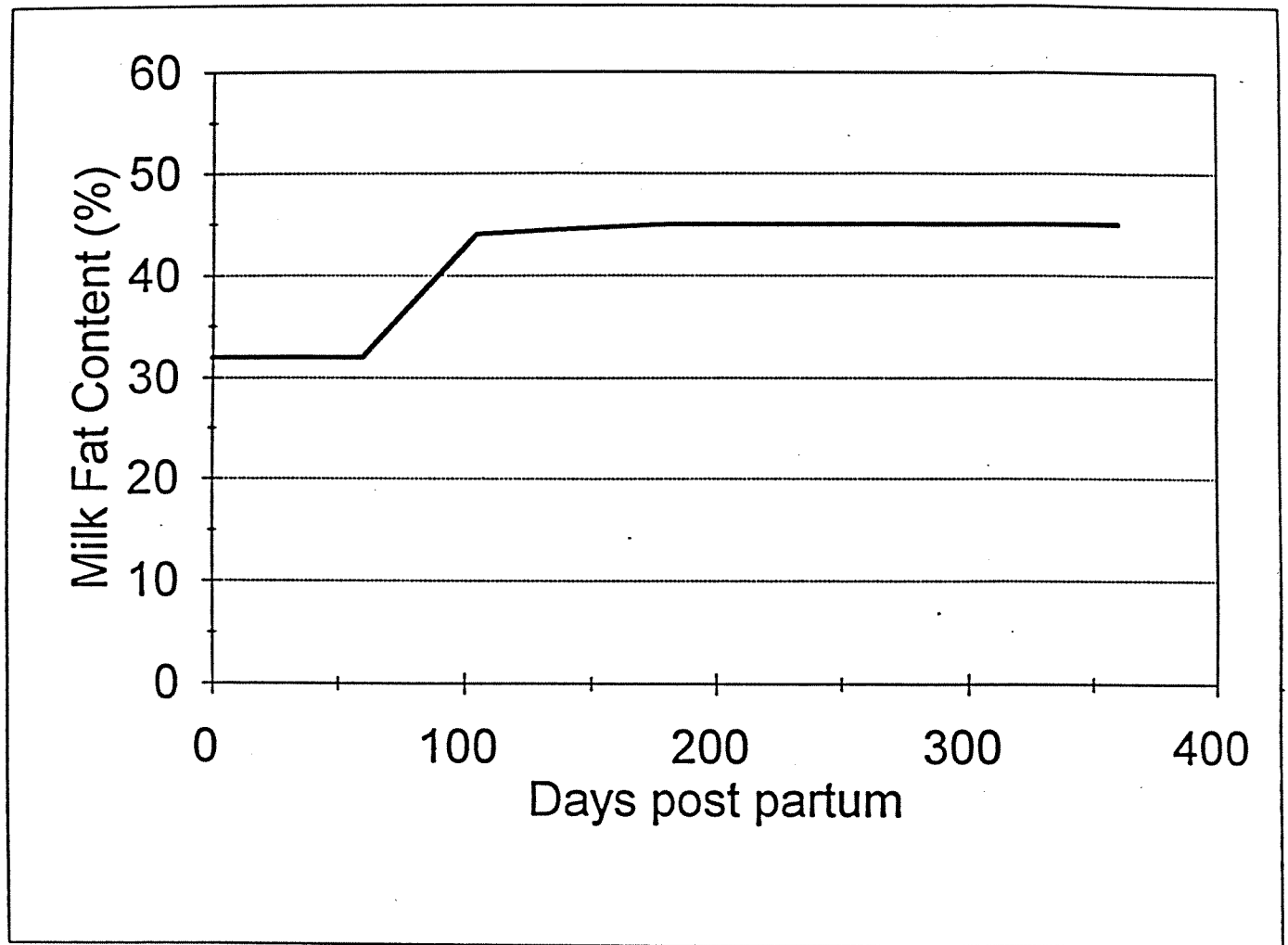
Figure 4-6. Milk fat content-days post partum relationship for sea lions.

blacksmith, Pacific mackerel, Octopus sp., nail squid, and senorita. These species are supplemented by 40 additional species which occur less frequently in the sea lion diet. Sea lions observed from Sealab II in 61 meters of water off La Jolla, California fed on anchovy, jack mackerel, white croaker and squid (Clarke *et al.* 1967).

p,p'DDE and PCB data are available for four of the predominant prey species: rockfish, mackerel, blacksmith, and octopus. Data are also available for three minor prey species: kelp bass, white croaker and sand-dab. Of the seven prey species for which data exist, kelp bass and white croaker were collected most frequently. To minimize bias to these species, mean values for each of the seven prey species were computed by year and the average and range of these means were compared to the prey concentrations required to produce the p,p'DDE and PCB levels observed in the sea lions. In general, white croaker are at the upper limit of the range, reflecting their closer association with the sediment than the other prey species. These data were used as the basis for comparison to the model.

4.5 MODEL RESULTS

4.5.1 General

The data and assumptions described in the preceding sections provide best estimates of the processes contributing to uptake and loss of p,p'DDE and PCBs. Together in the framework of the model they define the relationship between dietary contaminant concentrations and sea lion contaminant concentration. As discussed in the Introduction, the model has been developed to provide a means of estimating the historical concentrations of p,p'DDE and PCB in the diet of female sea lions from the Channel Islands.

The model indicates that a contaminated diet results in contamination within the female sea lion that varies greatly over its lifespan. This pattern is illustrated on Figure 4-7 which shows the sea lion whole-body p,p'DDE concentration (ug/g wet) in relation to age when the sea lion is at steady-state with a dietary concentration of 1 ug/g wet. Concentration increases rapidly during the first seven months of life peaking at about 74 ug/g. This increase results from the transfer of p,p'DDE from the mother by way of lactation. Once the pup begins to forage and decrease its ingestion of milk the p,p'DDE concentration decreases rapidly. This decrease occurs because of a decrease in dietary contamination and rapid rates of growth dilution and excretion. By about age three the animal has eliminated the p,p'DDE taken in during nursing and is now responding only to the contamination in its diet of fish. Concentration then
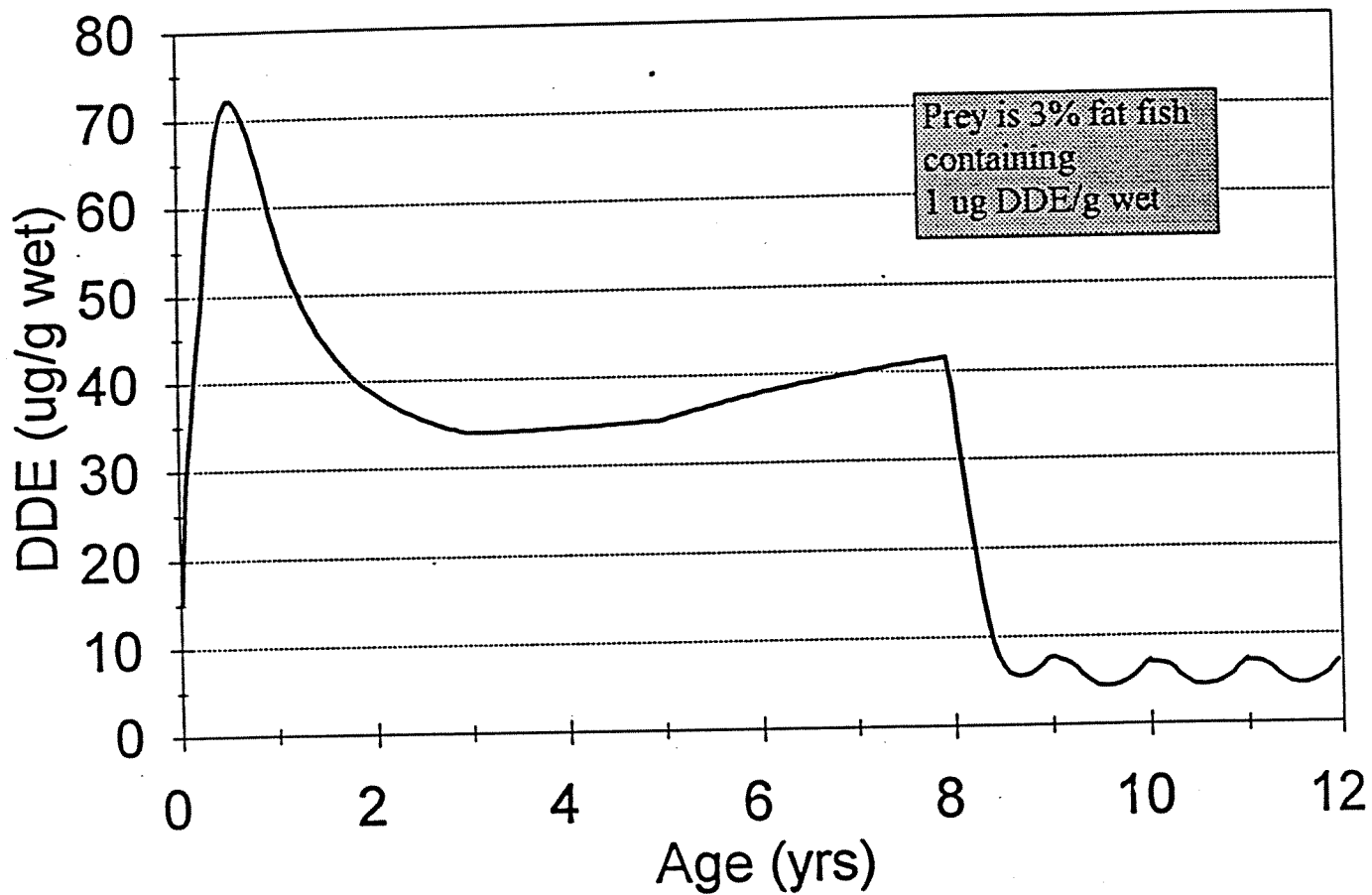
Figure 4-7. Predicted steady-state whole body wet weight p,p'DDE concentrations in female sea lions in relation to age for animals eating prey containing 1 μg DDE/g wet.

increases slowly to age eight. At this age the female begins her first lactation cycle following her first full-term parturition. The p,p'DDE concentration drops precipitously. During lactation the female is producing milk at a rate equal to about 1 percent of her body weight per day. Because the fat content of the milk is slightly higher than her body fat content, the p,p'DDE concentration in the milk is slightly higher than her body concentration. Thus, lactation results in a p,p'DDE loss rate of greater than 0.01/d. This loss persists for the remainder of her life because of the 11 month lactation cycle and the annual parturition. The rate of loss varies over the year as the rate of milk production varies, causing the sinusoidal variation in body burden evident on Figure 4-7. The pattern in the reproducing female would be somewhat different if she did not have a full-term parturition each year. In a year that she did not lactate her concentration would increase beyond that shown in the figure. The importance of lactation as a sink of persistent organochlorines in mothers and a source to pups has been demonstrated for grey seals (Addison and Brodic 1977), northern elephant seals (Newman 1991) and striped dolphins (Tanabe *et al.* 1981).

Thus, the female sea lion exhibits three distinct levels of contamination over her life span. The highest level occurs as a pup. An intermediate level is experienced as a juvenile. The lowest level occurs as an adult. Similar contamination patterns have been observed in laboratory studies. For example, a study of PCB concentrations in female and newborn rabbits following a single PCB dose administered during pregnancy (Montesissa *et al.* 1992) found a rapid increase in PCB concentration in the newborns during the time of nursing followed by a rapid concentration decline after nursing ceased (Figure 4-8). The concentrations in dams declined by a third during 10 days of nursing and then by only about 10 percent during the next thirty days.

The dietary p,p'DDE and PCB concentrations of the Channel Islands females were estimated by fitting the computed concentration profile to the data collected in 1970, 1972 and 1991. A continuous simulation extending from the late 1950's to 1991 was conducted. For the period up to 1970 the prey p,p'DDE and PCB concentrations were assumed to be constant. The focus of this part of the calculation was not to accurately estimate the temporal profile of prey concentration, but to provide best fit estimates of the concentrations of all ages of sea lion in 1970. For the period from 1970 to 1991 a temporal profile of prey concentration was established by attempting to reproduce the sea lion concentrations observed in 1972 and 1991. The only constraint placed on this pattern was that it be consistent with our observation that water, sediment and fish concentrations have remained relatively constant since the mid 1980s.

Figure 4-8. Response in female and newborn white rabbits to single PCB dose administered during pregnancy.
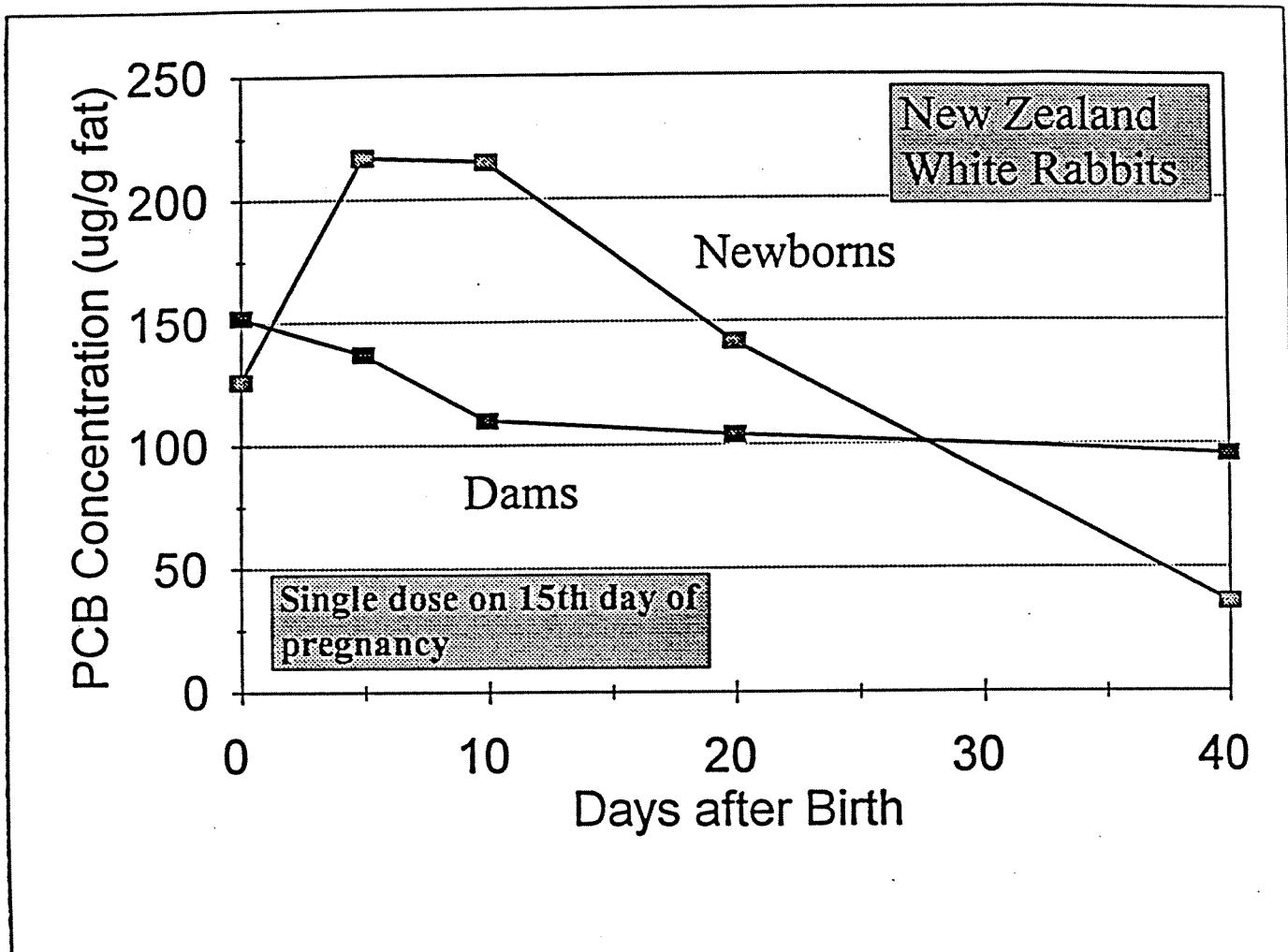
Ref: Montesissa et al., 1992, Pharm. Toxicol. 71:139-143

4.5.2 p,p'DDE and PCBs

The comparison of computed and observed sea lion p,p'DDE and PCB concentrations are presented on Figures 4-9 and 4-10. Using the prey contaminant profiles shown on Figure 4-11 the model fit both the general pattern of the observed concentration-age profiles and the observed concentration-time profiles. For both contaminants the computed concentrations are typically within a factor of two of the observed values. A few data points do differ more than a factor of two from the computed values, but, as discussed in the Section 2, these data may be from animals whose rearing pattern differs from that assumed in the model.

The prey contaminant profiles indicated by the model were compared to the concentration profiles observed in sea lion prey species. The observed values were grouped within three broad regions: Palos Verdes, North of Palos Verdes and Santa Catalina Island based on the model segments shown on Figures 2-3 and 2-4. The Palos Verdes region includes the area from segments 3 to 10. The North region includes segments 1, 2 and 13. Most of the data in this region is from segment 2 (upper Santa Monica Bay). Because foraging trips last several days and may cover several hundred kilometers, the females are capable of taking prey from a wide area and they may be exposed to contaminants at levels equivalent to those in any one or all three of these regions. The comparisons between the data and the required prey concentrations indicated by the model are shown on Figure 4-12 for p,p'DDE and Figure 4-13 for PCBs. Solid and dashed lines are shown to indicate prey concentrations necessary to achieve 1991 sea lion contaminant concentrations reported by Old GERG and New GERG, respectively. The concentrations observed at a low-level contaminant site (Santa Catalina Island) are insufficient to account for the p,p'DDE and PCB concentrations measured in the female sea lions. By contrast, the prey contaminant levels in the other two regions are sufficient to account for the measured concentrations. Although we cannot estimate the pattern of feeding resulting in the required prey contaminant level (e.g., feeding across the concentration gradient between the outfall and San Miguel Island versus feeding consistently in the Santa Monica Shelf area, i.e., the North region), we do know that the average sea lion prey had elevated p,p'DDE concentrations. A similar pattern is seen for PCBs although the association with elevated concentrations is less clear. Because, 1) adult females rearing pups spend most of the year foraging from San Miguel Island (a consequence of a 9 to 10 month nursing commitment) and, 2) the areas of observed elevated concentrations within foraging range of the island (i.e., the North region and the Palos Verdes region) are within the area effected by the Whites Point outfall and the Palos Verdes sediment plume the sea lion p,p'DDE and PCBs most probably originated from the Palos Verdes Shelf.
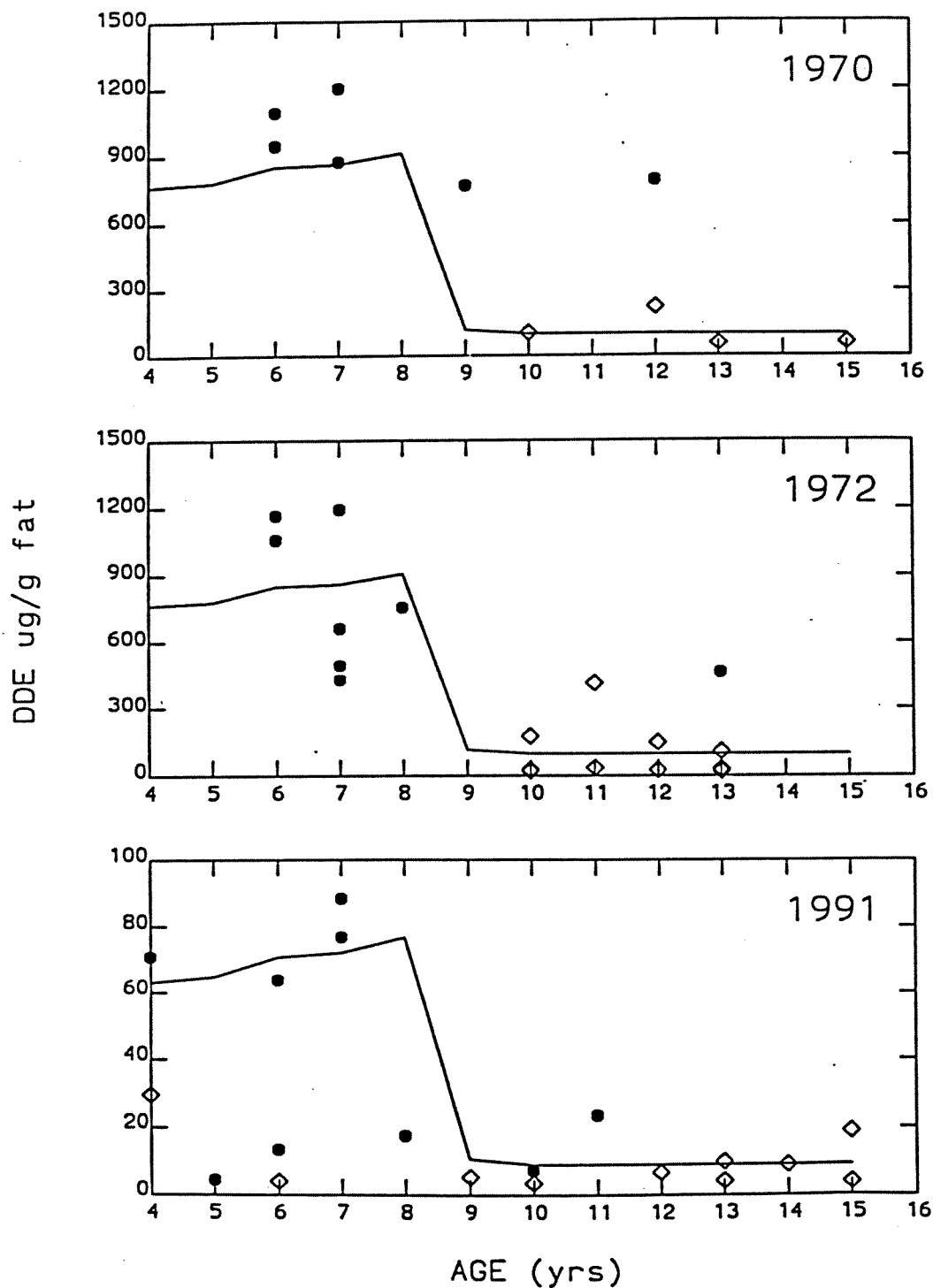
Figure 4-9. Comparison of computed (lines) and observed (symbols) p,p'DDE concentrations in female sea lions from San Miguel Island. Solid circles are premature parturient animals. Open diamonds are full-term parturient animals.
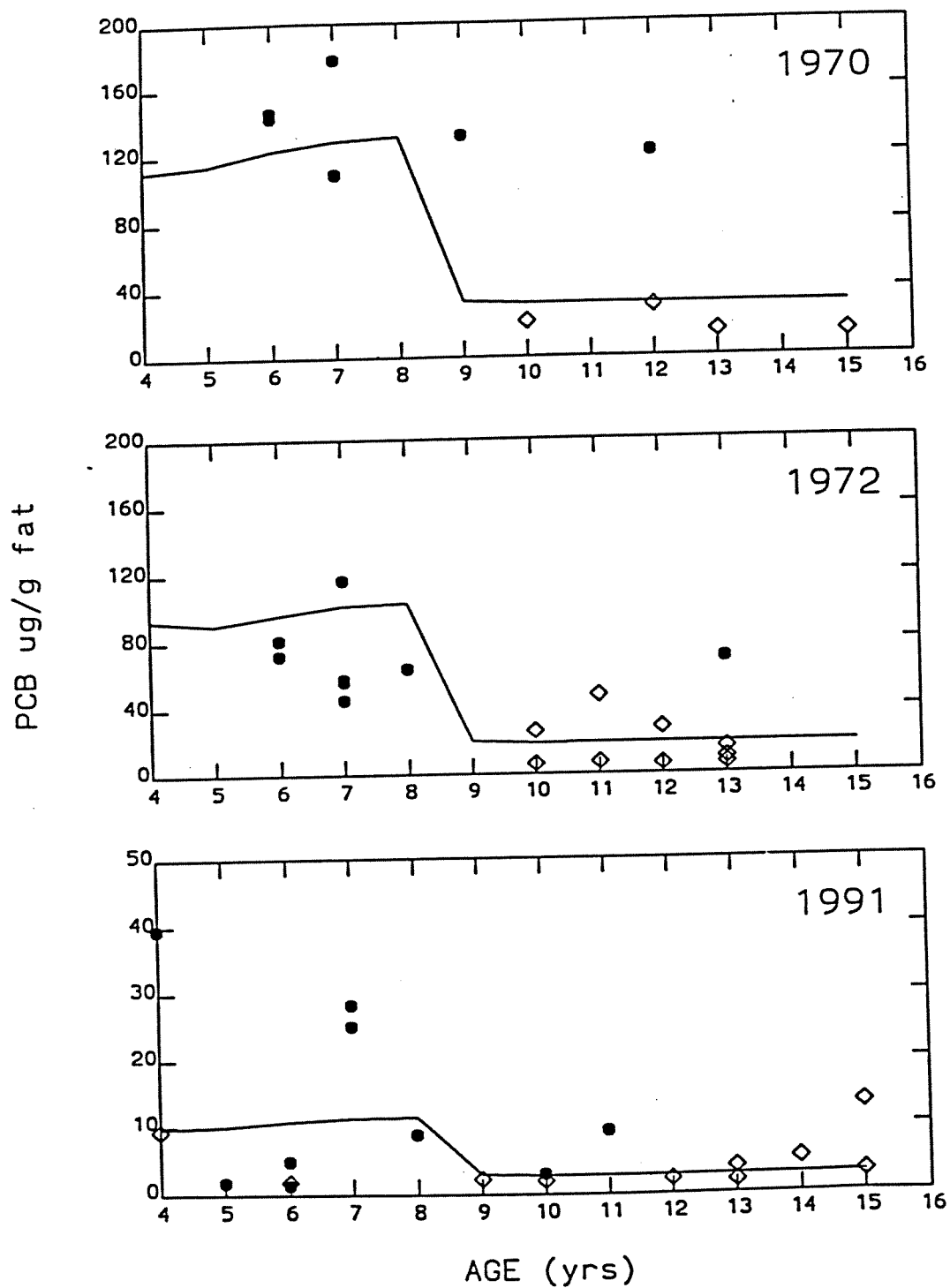
Figure 4-10. Comparison of computed (lines) and observed (symbols) PCB concentrations in female sea lions from San Miguel Island. Solid circles are premature parturient animals. Open diamonds are full-term parturient animals.
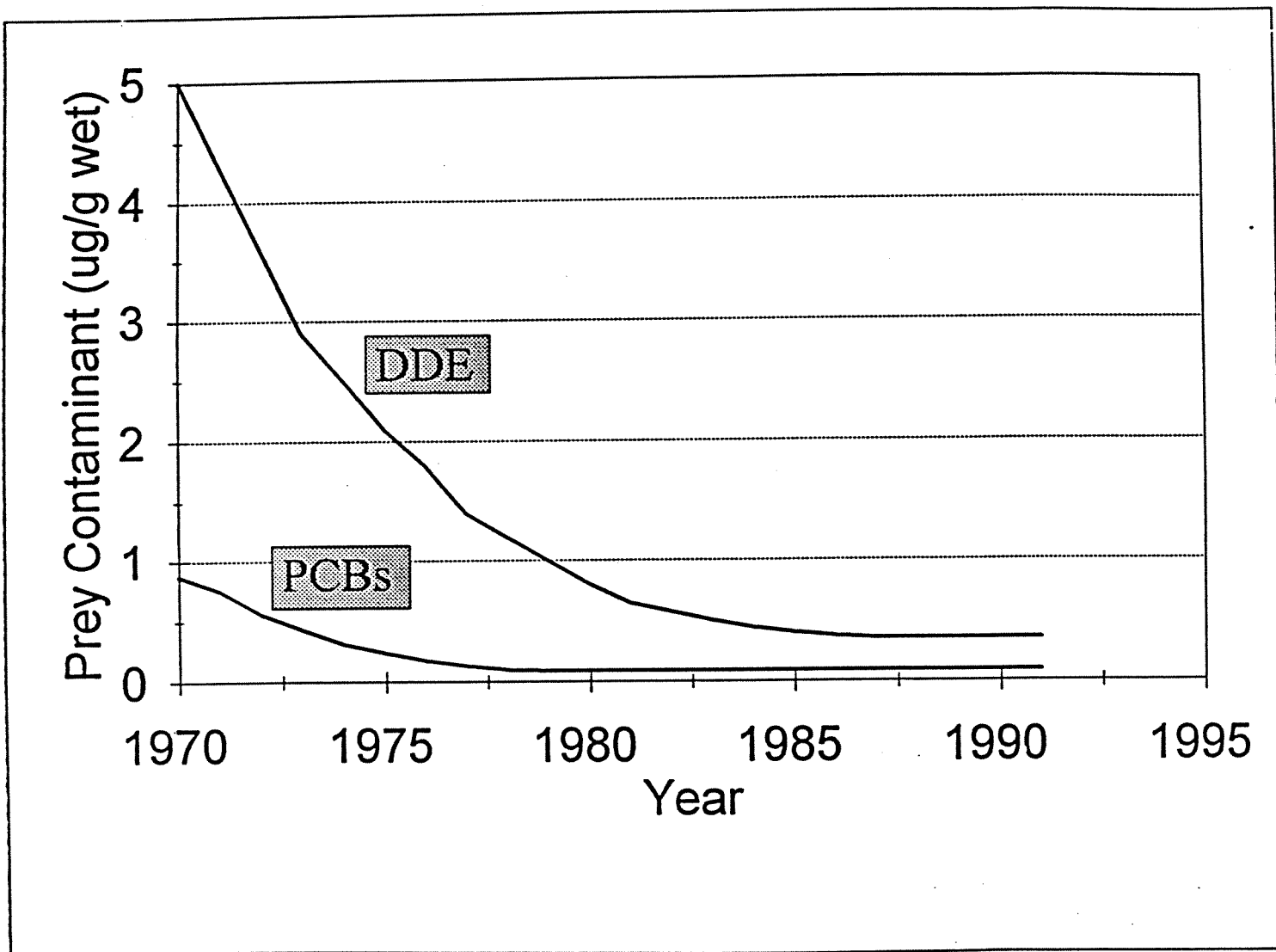
Figure 4-11. Temporal profile of the p,p'DDE and PCB concentrations required to be in female sea lion prey to achieve the p,p'DDE and PCB concentrations observed in San Miguel Island sea lions.

CONTAMINATION IN FISH PREY
OF THE CALIFORNIA SEA LION
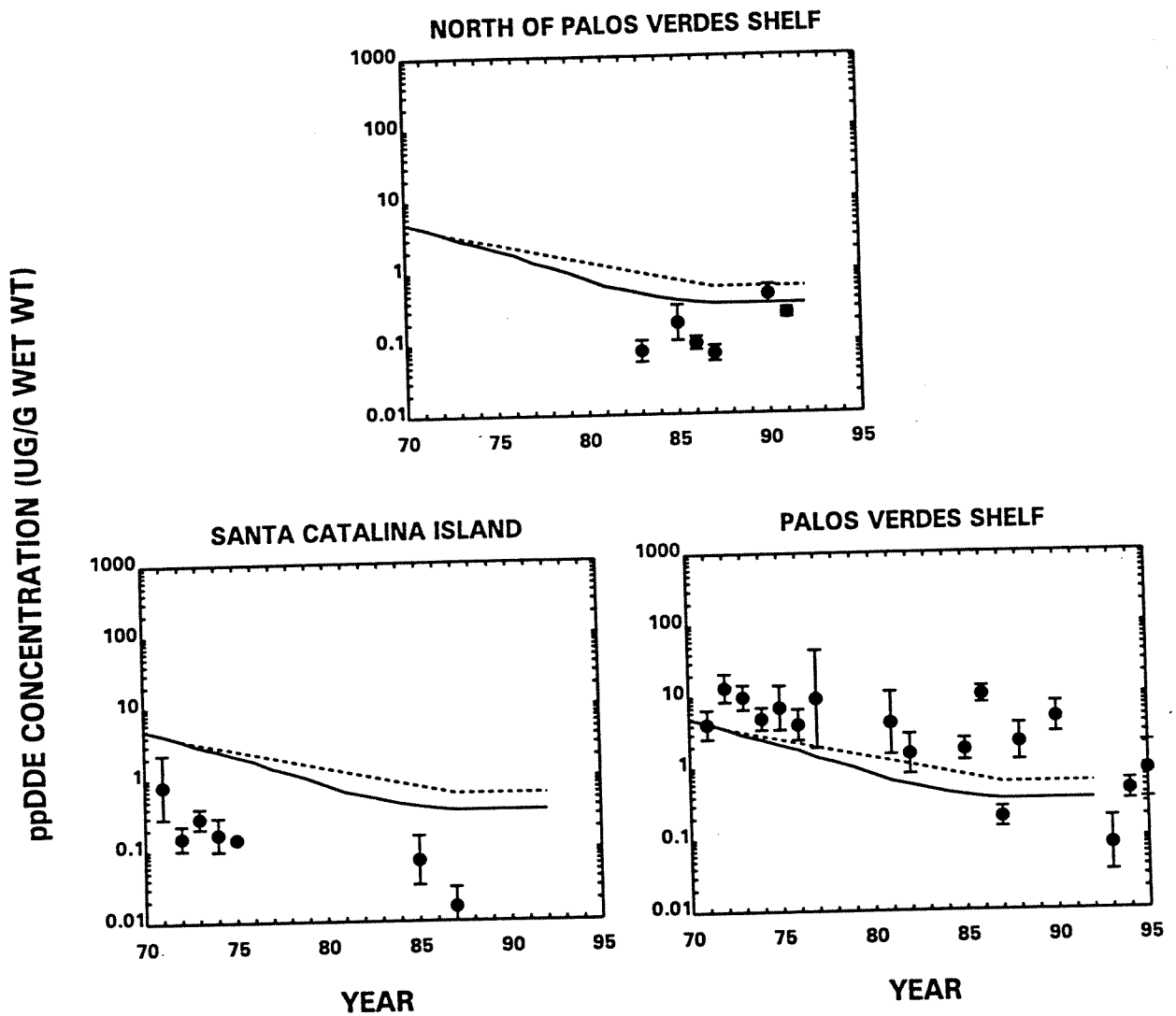
Figure 4-12. Comparison of computed prey p,p'DDE concentration profiles (lines) and concentration profiles observed in sea lion prey species for three regions of the Southern California Bight (symbols). Solid lines indicate old GERG data; dashed lines indicate new GERG data. Data are arithemtic mean +/- 2 standard errors of the mean.

CONTAMINATION IN FISH PREY
OF THE CALIFORNIA SEA LION
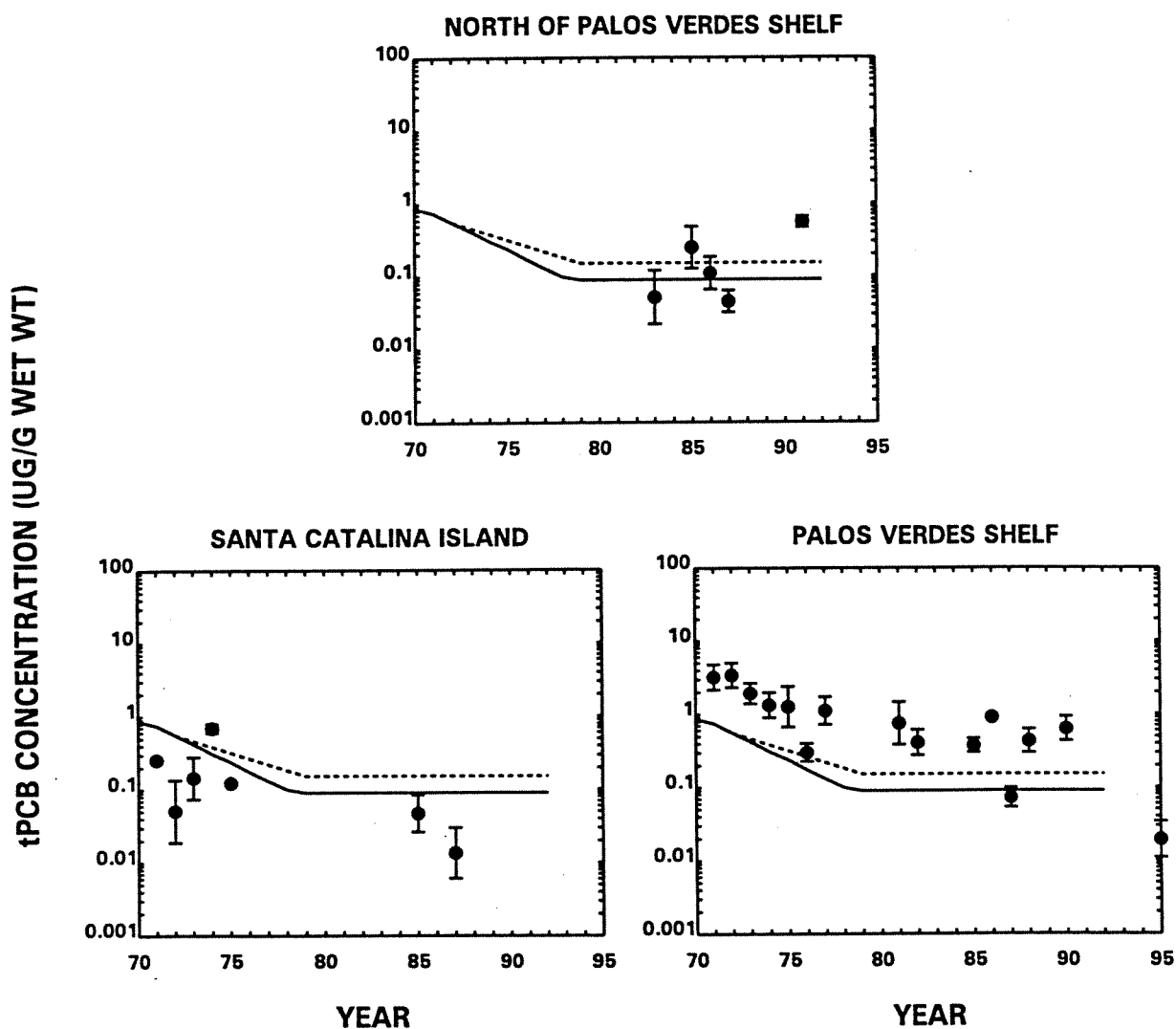
Figure 4-13. Comparison of computed prey PCB concentration profiles (lines) and concentration profiles observed in sea lion prey species for three regions of the Southern California Bight (symbols) Solid lines indicate old GERG data; dashed lines indicate new GERG data. Data are arithemtic mean +/- 2 standard errors of the mean.

The relationships between p,p'DDE and PCB concentrations in female sea lions and their prey are shown in generalized form for continuous exposures on Figures 4-14 and 4-15. These figures illustrate the linearity of the relationship for a single aged animal. For both contaminants, concentrations decrease by about a factor of 10 from pups to reproducing adults.

4.5.3 Sensitivity Analysis

The values used to define a number of the biological and toxicokinetic processes in the model represent best estimates that are subject to unknown uncertainty. Of particular note are growth rate, lipid content, milk production rate, period of maximum milk production, contaminant:food energy assimilation efficiency ratio and contaminant excretion rate. Errors in any of these parameters affect the results by altering the relationship between the concentration in sea lions and in their diet. Because the model is linear to prey concentration, changes in sea lion concentrations resulting from parameter value changes imply equivalent opposite changes in the prey concentration necessary to reproduce the observed sea lion concentrations.

To assess the importance of parameter uncertainty, changes in computed p,p'DDE concentrations in 4, 7 and 11 year old female sea lions were calculated for discrete changes in the above parameters. Excretion rate was not included in this analysis because the sensitivity of the model to this parameter was presented in Section 4.2.3. Briefly, lactating adults are relatively insensitive to the excretion rate because of the dominance of lactation loss, whereas concentrations in juveniles and non-lactating adults are affected by changes in the excretion rate. Because of this, the value of excretion was determined from the concentration ratios between the two groups.

The results of the sensitivity analysis are presented in Table 4-3 as the percentage change in p,p'DDE concentration occurring from the change in the parameter value. It is evident that the parameter most affecting the model is the contaminant assimilation efficiency. Computed sea lion p,p'DDE (and PCB) concentrations are directly proportional to its value. Thus the 33 percent change from 0.75 to 1.0 or 0.5 results in a 33 percent change in concentrations. The model is much less sensitive to changes in the biological parameters. Reducing growth rate to match the DeLong data causes a small increase in concentration in the four year old because growth is a important loss mechanism at this age. The slight decline in concentration in the older animals results from a reduction in ingestion rate associated with reduced growth that is not fully compensated for by the reduced growth dilution.
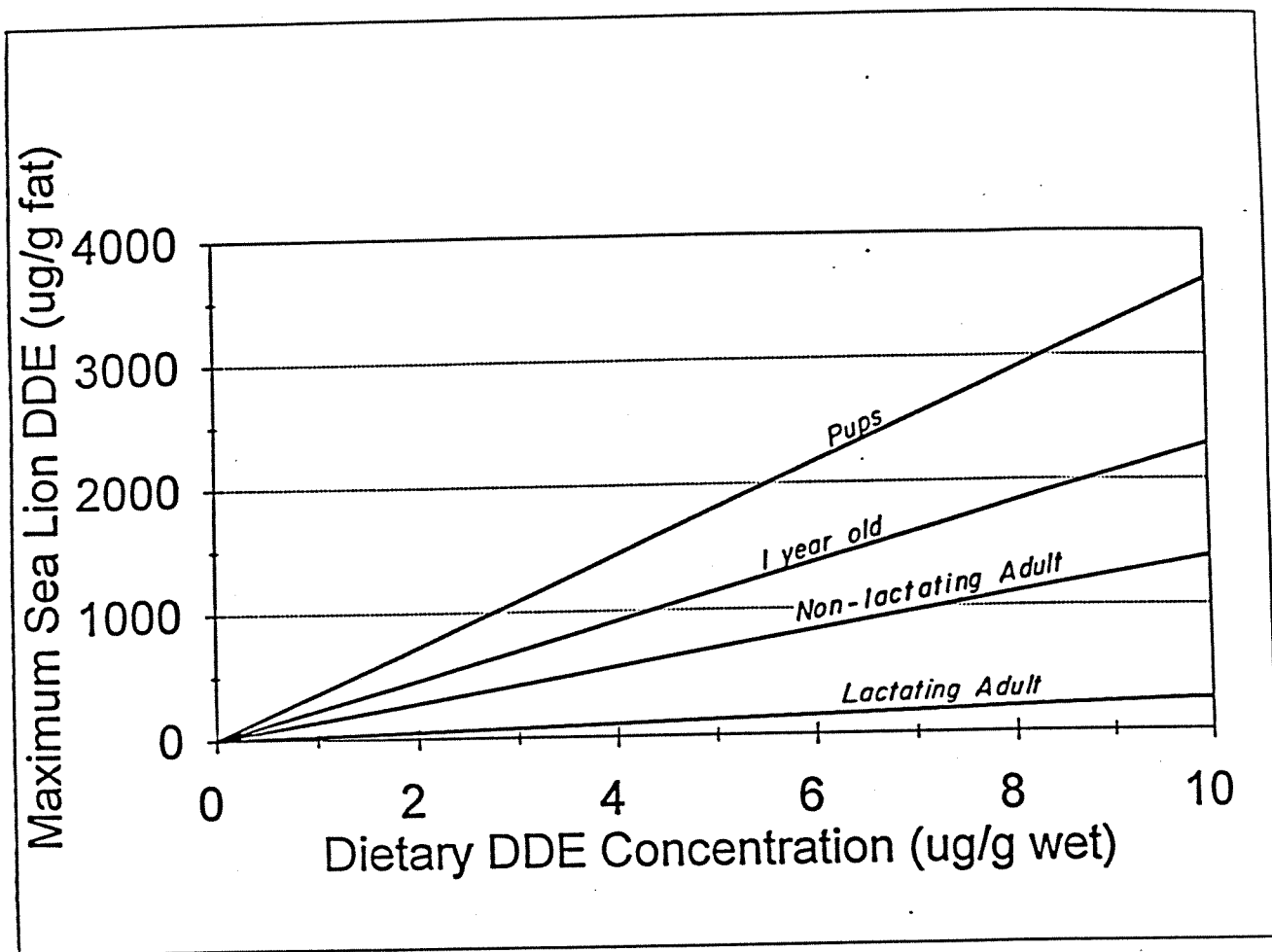
Figure 4-14. Computed maximum p,p'DDE concentration in female sea lions of various ages in relation to dietary p,p'DDE concentrations. The dietary concentrations represent long-term average exposure concentrations.
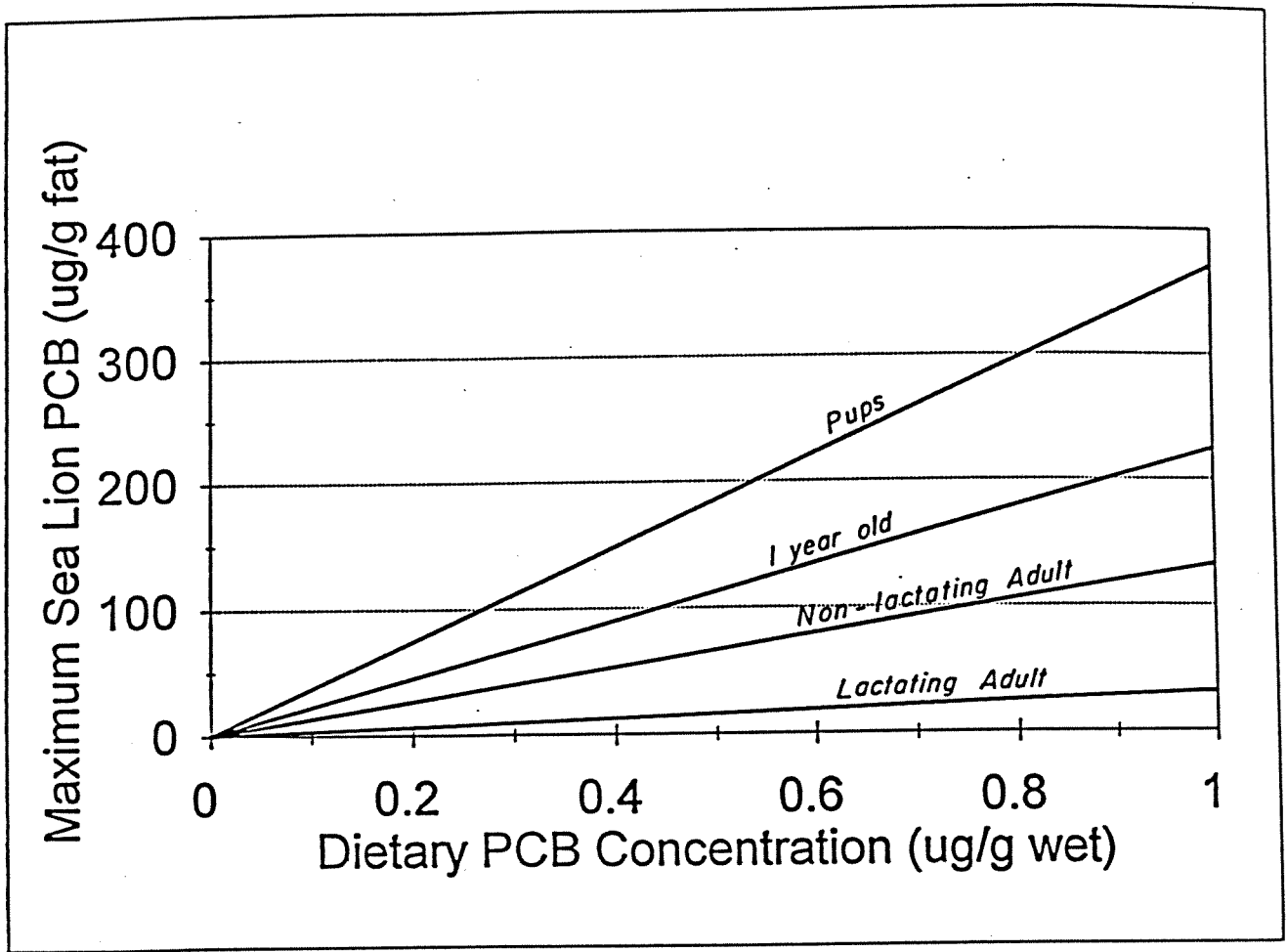
Figure 4-15. Computed maximum PCB concentrations in female sea lions of various ages in relation to dietary PCB concentrations. The dietary concentrations represent long-term average exposure concentrations.

Table 4-3. Changes in the Computed Steady-State Concentrations of Sea Lion p,p'DDE that Result from Changes in the Values of Various Biological and Toxicokinetic Parameters

Process	Alteration	% Change in Concentration		
		4 Year Old	7 Year Old	11 Year Old
Growth	reduce growth to match DeLong (unpublished) data shown in Figure 4-2	12	-4	-4
Whole-body lipid content	increase body fat of animals 2 years or older from 30 to 40 percent	-8	-1	-8
Whole-body lipid content	decease body fat of animals 2 years or older from 30 to 20 percent	1	1	13
Lactation	decrease in milk production starts 5 months post partum rather than 6 months post partum	<1	<1	-6
Lactation	decrease in milk production starts 7 months post partum rather than 6 months post partum	<1	<1	4
Lactation	reduce maximum milk production from 1 kg/d to 0.8 kg/d	<1	<1	-7
Lactation	increase maximum milk production from 1 kg/d to 1.2 kg/d	<1	<1	9
Contaminant assimilation	decrease contaminant:food energy assimilation efficiency ratio from 0.75 to 0.5	-33	-33	-33
Contaminant assimilation	increase contaminant:food energy assimilation efficiency ratio from 0.75 to 1.0	33	33	33

Changing animal lipid content by a factor of 2 (20 to 40 percent) did not have a great effect on concentrations because the steady-state concentration per unit lipid weight is relatively independent of the absolute lipid content. This is so because the lactation and excretion loss rates change in proportion to whole-body lipid content. Thus, halving the lipid content doubles the loss rates and reduces whole-body concentration. The effect that is seen occurs for two reasons. First the total loss rate includes growth dilution which mitigates the proportionality between total loss rate and lipid content. Second, lipid content affects energy density, and thus ingestion rate.

Changes in the rate or duration of milk production affected only the lactating animals as exemplified by the 11 year old. The four and seven year old are not effected because their concentrations are derived from dietary exposure. Most of the contaminant they accumulated as nursing pups has been lost through a combination of growth dilution and excretion. The changes in the 11 year old are in approximate proportion to the magnitude of the milk production changes relative to total milk production.

In summary, the sensitivity analysis indicates that the uncertainty in the biological parameters imparts little uncertainty to the model. Even the compounded uncertainty of these parameters affects results by something less than a factor of two. Of greater significance is uncertainty in the contaminant assimilation efficiency. Uncertainty in this parameter causes equivalent uncertainty in the model. The uncertainty in this parameter is probably about a factor of two. The sensitivity of juveniles and non-lactating adults to the excretion rate implies uncertainty regarding the contaminant concentration in the prey of these animals. Just as their concentration may be altered by changes in excretion rate it may be altered by changes in prey contaminant concentration. By contrast, the unimportance of excretion for lactating adults means that the relationship between their contaminant concentration and that in their prey is less uncertain. In fact, because this relationship is sensitive only to processes that have been measured (metabolism and milk production) and the contaminant:food assimilation efficiency, its uncertainty is likely about a factor of 2, or so. Overlaying such a range on the predicted prey contaminant concentrations does not cause an overlap with the concentrations observed at Santa Catalina Island. Thus, uncertainty does not alter the conclusion that the sea lion prey must have had elevated contaminant concentrations.

SECTION 5

FOOD CHAIN MODELS OF BIRDS

A model framework that describes contaminant dynamics in birds was applied to bald eagles, peregrine falcons and double-crested cormorants of the Southern California Bight. Body composition, bioenergetics, dietary composition, contaminant levels in prey and contaminant toxicokinetic parameter values (assimilation efficiency, transfer to the eggs and excretion rate) were established for each species from published data and data collected as part of the Southern California Bight Damage Assessment. Because the models were mechanistically-based and were parameterized using species- and site-specific data, they are valuable tools for quantifying the relationship between exposure sources and contaminant levels in the eggs of the three species of interest.

5.1 MODEL STRUCTURE

The dynamics of contaminant accumulation in birds differs from that in mammals, because they do not lactate; thus, nursing is neither a loss mechanism for adults nor an exposure mechanism for young. The primary loss mechanisms are excretion and egg production. Model results for females are presented in this section, because the focus of the pathway work is the bird eggs.

A bird is represented as having two body compartments between which the contaminant moves: blood and lipid (Clark *et al.* 1987). The blood is the compartment into which contaminant enters from the gut and out of which contaminant is excreted. Contaminant is transferred between blood and body lipid, primarily fat stores. The lipid compartment is considered a "deep" compartment, because the contaminant accumulates in lipid, which limits its availability to excretory organs.

As with the models for fish and mammals, contaminant intake is calculated by multiplying contaminant levels in the prey (ppm wet weight prey) by a consumption rate (g wet weight prey/g wet weight bird whole body-day; hereafter gwet prey/gwet wb-day). The consumption rate is determined from a simple energy balance (see Appendix A for model equations).

One mechanism by which contaminant is lost from a bird is the production of eggs. Contaminant concentration in the egg (ppm lipid) is a weighted average of concentrations in dietary and body lipid (see Appendix 1).

The egg is produced over a period of a several days following these rules:

- Every day, dietary and body lipids are added to the growing yolk.
- Once yolk is laid down, it is isolated from other body compartments.
- During incubation, total mass of contaminant in the egg remains constant, even though lipid and water contents change.

5.2   DEVELOPMENT OF THE PEREGRINE FALCON MODEL

5.2.1   Growth and composition

Life history and growth rate.   Peregrine falcons reach their adult weight at fledging, after approximately 40 to 45 days (Ratcliffe 1993). Average adult weights of peregrine falcons are approximately 1000 g (female) and 680 g (male; Ratcliffe 1993).   Once birds reach their adult weights, the growth occurs as egg production. In the model, the first egg is laid at age two. Falcons can live to age 20 but probably do not live this long in nature (Ratcliffe 1993). The model considers birds up to 15 years of age.

Adult whole-body lipid content. The lipid content of the birds controls the excretion rate; a higher lipid content leads to a lower excretion rate. Estimates of whole-body lipid content that are available in the literature range from 0.05 to 0.20 g lipid/gwet (hereafter glip/gwet) for several species of birds (Table 5-1). A value of 0.075 glip/gwet is used for the adult peregrine falcon, based on a measured value of 0.05 for peregrine muscle (Cade *et al.* 1968) and a whole-body:muscle lipid ratio of 1.5 derived from the work of Norstrom *et al.* (1986b) for herring gulls (0.05 * 1.5 = 0.075).

Lipid fraction in peregrine falcons of the Southern California Bight is unlikely to vary substantially throughout the year, because the resource base is rich in the southern California Bight and temperature variation is small, and because these birds do not migrate long distances (Hunt 1994). Lipid content variation in nestlings is incorporated into the model (Table 5-2).

Table 5-1. Whole-Body Lipid Content in Birds
(g lipid/g(w) weight)

Species	Value	Notes	Reference
Herring gull	0.08 - 0.18	wild adults spring and summer, Great Lakes	Norstrom *et al.* 1986
American kestrels	0.05 - 0.12	carcass of captive female birds fed DDE	Wiemeyer *et al.* 1986
American kestrels	0.20	birds held captive for 1 month and fed DDE	Henny and Meeker 1981
Sparrowhawks	0.09	maximum value for birds found dead in Great Britain	Bogan and Newton 1977
Alaskan peregrine falcons	0.05	muscle, trapped birds	Cade *et al.* 1968
Double-crested cormorants	0.06	caloric contents of juveniles	Dunn 1975
Double-crested cormorants	0.07 (0.01-0.12)	wild adults, breeding period	Dale and Stromborg 1993

Table 5-2. Growth and Body Composition for the Peregrine Falcon

Stage	Time (days)	Weight (g(w))	Proportion Lipid
Yolk production[4]	(10 days)		
Egg laid	0 (early February)	46[1] (3.5/clutch)[3]	0.044[2]
Egg hatched	33	35	0.033[5]
Fledge	76	1000 (female) 680 (male)	0.075[2]

NOTES: All values from Ratcliffe 1993 unless noted.
[1]Southern California Bight Damage Assessment data.
[2]See discussion in text.
[3]Ratcliffe 1993.
[4]Roudybush *et al.* 1979
[5]Cade *et al.* 1968

Adult whole-body dry weight. Fraction dry weight affects the energy content of the bird and is used to estimate consumption rates. Fraction dry weight averaged 0.35 g dry/gwet in American kestrels (Wiemeyer *et al.* 1986), and a value of 0.34 was estimated for Alaskan peregrines from the data of Cade *et al.* (1968). A value of 0.35 was used in the peregrine falcon model.

Egg lipid content. The lipid content of eggs at laying is required to estimate the concentrations of contaminants in eggs (see Appendix 1). Lipid contents were measured in eggs of peregrine falcons collected in the Southern California Bight in 1992, as part of the Southern California Bight Damage Assessment. The average fresh weight-corrected value (+/- one standard deviation) was 0.036 glip/gwet +/- 0.010 (n=16).

However, the age of the eggs collected as part of the Southern California Bight Damage Assessment was variable, and several had undergone some development. During development, lipid content is expected to decline as it is used for energy by the developing embryo. Consistent with this, the lipid contents of the Southern California Bight eggs were lower than values found in the literature. For example, the proportion lipid of fresh American kestrel (Falco sparverius) eggs averaged 0.051 (range 0.039 to 0.061; Wiemeyer et al. 1986). Cade et al. (1968) measured an average lipid content of 0.044 in peregrine falcon eggs (computed from lipid contents reported as a fraction of dry weight using a value of 0.26 gdry/gwet). The age of the eggs was unknown. This value was used in the model as the lipid content at the time of laying. As a check on this value, the ratio of lipid contents between egg/whole body (0.044/0.075 = 0.59) is consistent with measurements made in captive American kestrels (average = 0.65, range = 0.45 to 0.95; Wiemeyer et al. 1986).

5.2.2 Metabolic Rate

Contaminant dose (μg/gwet-day) is computed by multiplying concentration in prey (μg/gwet) by the rate of food consumption (gwet/gwet-day). Consumption rate is estimated from the sum of growth + metabolic rates. Thus, estimates of metabolic rate are necessary.

Field metabolic rates (FMR) have been measured in 62 species of birds ranging in size from 4.5 to 88,000 g (Williams et al. 1993). Based on these data, Williams et al. (1993) developed a relationship with body weight:

$$\text{FMR} = 9.57 \ W^{-0.311}, \quad n=62, \quad r^2=0.93 \tag{5-1}$$

with FMR in kJ/g-d and W in grams. The adult peregrine falcon metabolic rate based on Equation 5-1 is 1.12 kJ/gwet-d.

This estimate of total metabolic rate was checked by comparison to reported resting metabolic rates; it should be 2 to 3 times greater than the resting metabolic rate (Walsberg 1993). Consistent with this, a bioenergetics model developed by Koplin *et al.* (1980, using a model framework developed by Kendeigh *et al.* 1977) yields a resting metabolic rate of 0.58 for a peregrine falcon. In addition, an average resting metabolic rate of 0.25 kJ/g-day can be calculated from the measurements of 11 species of falconiformes in the thermoneutral zone performed by Wasser (1986). This value is more than a factor of 3 lower than the estimated total metabolic rate, which is consistent with the fact that the laboratory measurements were all collected in the thermoneutral zone, that is, the birds were not thermally stressed.

5.2.3 Feeding Habits

The peregrine falcon feeds exclusively on birds (Kiff 1980). Over 50 species of birds have been observed as prey on the California Islands (Hunt 1994). Table 5-3 contains estimates of the proportions of several groups of birds in the diet of Channel Islands peregrine falcons (Hunt 1994). The field observations of diet were conducted in 1992 and 1993, during January and February ("winter diet") and during April and May ("spring diet"). The dietary compositions for the other times of year were slightly modified from the Spring diet using information provided by Grainger Hunt (author of Hunt 1994), based upon his general knowledge of peregrine falcon natural history and his particular knowledge of the population of the Southern California Bight. The annual average dietary composition is reported in Table 5-3.

Table 5-3. Dietary Composition, Contaminant Levels and Body Composition for Prey
of the Peregrine Falcon

	Proportion of diet on a wet-weight basis	body composition of prev		proportion of diet on an energy basis	contaminant levels (ppm wet weight)	
		fraction dry	fraction lipid		p,p'DDE	PCB
Western gulls	0.11	0.35	0.05	0.096	4.0	1.3
California gulls	0.092	0.39	0.08	0.094	2.9	0.90
Heermann's gulls	0.026	0.33	0.04	0.022	2.9	0.90
Bonaparte's gulls	0.022	0.38	0.06	0.022	2.9	0.90
Cassin's Auklets	0.17	0.38	0.10	0.18	2.2	0.42
Other waterbirds	0.26	0.38	0.06	0.26	1.2	0.39
Land birds - resident	0.19	0.38	0.06	0.19	0	0.0
Land birds - migratory	0.13	0.38	0.06	0.13	0.33	0.26
SUM	1					

Notes: Arithmetic means used.
Proportion of dietary and energy basis = proportion in a wet-weight basis x energy content, standardized to a total of 1.0.
Average annual diet calculated from the seasonal measurements of Hunt (1994), assuming that a slightly modified spring diet continued through the rest of the year.

The group "other water birds" consists of grebes, shearwaters, waterfowl, shorebirds and phalaropes. These birds are grouped together, because (1) all of these species feed on food associated with the water, and (2) there are few data on contaminant levels in any of these species.

The energy requirements of the predator (kJ/g-day) govern the prey consumption rate. Field-measured dietary preferences must be converted to units of energy. The energy content of each prey type is estimated from its dry weight and lipid content (see Equation A-8 in Appendix A). Lipid contents measured in gull whole-body samples collected in the Southern California Bight as part of the Southern California Bight Damage Assessment were used to estimate average lipid contents in several gull species: 0.05 for western gull, 0.076 for California gull, and 0.042 for Heerman's gull. The whole-body fraction lipid of Cassin's auklet was set equal to 0.1 to be consistent with the fraction lipid measured in the eggs and the assumption of equal lipid contents in eggs and whole bodies. For other species of gulls and for other waterbirds, the overall average for gulls was used (0.06).

The average fraction dry weight of whole birds collected in the Southern California Bight were 0.35 for western gulls, 0.39 for California gulls, 0.33 for Heermann's gull. Fraction dry weight averaged 0.37 in herring gulls from the Great Lakes (Norstrom et al. 1986b). Mahoney

and Jehl (1984) found an average value of 0.38 amongst a variety of marine birds from southern California and Mono Lake, California. Ellis and Jehl (1987) found that dry mass was 0.30 to 0.35 of lean body mass in several species of birds; this measurement differs from the other cited values by fat content. A value of 0.38 was used for all avian prey for which site-specific data were not available.

The dietary preferences of the peregrine falcon, on a numerical and on an energy basis, are presented in Table 5-3. The energy-based preferences were used in the model.

5.2.4 Contaminant Levels in Prey

Concentrations of contaminants in prey of the peregrine falcon were estimated using the Southern California Bight Damage Assessment database. Values for birds collected on San Miguel, Prince, Santa Cruz, Santa Rosa and Anacapa Islands were used, except for other waterbirds as described below. Whole-body concentration values for each species were combined into a single average.

The gull contaminant database. In order to estimate prey concentrations for the peregrine falcon and bald eagle, a database of concentration values in gulls of the Southern California Bight was constructed. Measured whole-body concentrations were used directly. Concentrations measured in breast tissue and eggs were converted to whole-body. Western gull egg data were converted to whole-body concentrations using the following equation:

$$\frac{\mu g\ DDE}{gwet_{egg}} \times \frac{\left(\dfrac{0.0447\ glip/gwet_{wb}}{.0639\ glip/gwet_{egg}}\right)}{\left(0.537\ \dfrac{\mu g/glip_{egg}}{\mu g/glip_{wb}}\right)} = \frac{\mu g\ DDE}{gwet_{wb}} \tag{5-2}$$

The lipid contents in whole body (0.0447 glip/gwet) and egg (0.0639 glip/gwet) samples for western gulls collected on Santa Catalina Island as part of the Southern California Bight Damage Assessment were used. The factor of 0.537 used for egg/whole body lipid-based contaminant concentration ratio is discussed in the Section 5.5.

The conversion from breast to whole body required two conversion factors, both of which were derived from the studies of Norstrom et al. (1986a and 1986b) amd Braune and Norstrom (1989). A value of 0.66 was used for the ratio of lipid fractions between breast and whole body, and a value of 0.85 was used for the ratio of lipid-based contaminant levels between the breast and the whole body.

The calculated and measured gull whole-body concentrations are summarized in Table 5-4. This table includes all gull samples collected as part of the Southern California Bight Damage Assessment, as well as samples of western gulls taken from Santa Catalina Island in 1986 and 1989 (David Garcelon, Institute for Wildlife Studies, Arcata, California, unpublished). These data were used to estimate contaminant levels in prey of both the peregrine falcon and the bald eagle.

One check of the calculated whole-body concentrations can be made for western gulls on Santa Catalina Island. The measured whole-body p,p'DDE concentration is 8.55 +/-5.05 (n=10), and the value calculated from egg data is 8.06 +/- 3.22 (n=12), which do not differ significantly. This supports the validity of the method of conversion between egg and whole body.

Some patterns are evident in the data. Concentrations are generally lowest on Anacapa, Santa Cruz and Santa Rosa Islands; about 1.5 ppm calculated whole body in western gulls, 2.9 ppm measured whole body in California gulls and about 6.9 ppm measured whole body in western gulls. San Miguel and close by Prince Island have higher concentrations; about 11 and 3.6 ppm calculated whole body, respectively for western gulls. These higher values may reflect predation on sea lion carcasses by the birds on these islands. Concentrations on Santa Barbara, Santa Catalina and San Clemente are similar to these values; calculated concentrations in western gulls of 3.9 to 8.3 ppm, measured values in California gulls averaging 7.8 on Santa Barbara and a single western gull measured value on Santa Barbara of 18.7 ppm. This last value is an outlier; the fraction of dry weight was high (0.84 gdry/gwet). Heermann's gulls were sampled only at one location (Santa Catalina). These data indicate higher concentrations than the other gulls; 20.4 ppm versus 8.1 for western gulls and 2.7 for California gulls. Because only two gulls were sampled, the significance of this difference is uncertain.

The PCB concentrations in the gulls indicate an identical pattern to that found for p,p'DDE, although levels are about three to six times lower.

Table 5-4. Calculated and Measured Whole Body Contaminant Concentrations in Gulls of the Southern California Bight

Species	Location	Tissue	Year	No. Obs.	Calculated and Measured Whole Body Concentration (ppm wet)					
					p,p'-DDE		Total DDT		PCBs	
					Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.
L. occ.	Anacapa	egg	1992	12	1.53	0.52	1.57	0.53	0.45	0.13
L. occ.	Prince	egg	1992	2	3.61	1.06	3.67	1.05	0.87	0.08
L. occ.	S. Miguel	egg	1992	8	11.04	10.83	11.13	10.85	2.78	2.92
L. occ.	S. Cruz	egg	1992	12	1.48	0.38	1.57	0.45	0.62	0.10
L. occ.	S. Catalina	egg	1992	12	8.06	3.22	8.28	3.23	2.01	0.59
L. occ.	S. Barbara	egg	1992	11	3.90	1.04	3.98	1.05	0.70	0.25
L. occ.	S. Clemente	egg	1992	13	8.34	5.91	8.46	5.95	2.04	0.99
L. occ.	S. Catalina	whole body	1992	10	8.55	5.05	8.62	5.08	1.81	0.99
L. occ.	S. Rosa	whole body	1993	1	6.85	-	6.94	-	1.24	-
L. occ.	S. Barbara[1]	whole body	1993	1	18.67	-	18.90	-	3.42	-
L. hee.	S. Catalina	whole body	1992	2	20.39	11.10	20.75	11.24	3.73	1.68
L. cal.	S. Catalina	whole body	1993	4	2.68	2.44	2.72	2.44	0.68	0.28
L. cal.	S. Barbara	whole body	1993	2	7.78	7.17	7.82	7.18	1.49	0.30
L. cal.	S. Cruz	whole body	1993	8	2.85	2.34	2.92	2.36	0.90	0.92
L. cal.	S. Cruz[2]	tissue	1993	1	1.24	-	1.30	-	0.54	-
L. occ.	S. Catalina	breast	1985	7	4.07	3.25	-	-	-	-
L. occ.	S. Catalina	breast	1989	9	11.53	2.06	-	-	3.05	0.73
L. hee.	S. Catalina	breast	1985	7	5.65	5.16	-	-	-	-
L. hee.	S. Catalina	breast	1989	10	4.82	1.73	-	-	1.47	0.47
L. cal.	S. Catalina	breast	1985	8	4.67	7.73	-	-	-	-

Notes:
L.hee = Heermann's gull, L.cal = California gull, L.occ = western gull
Contaminant levels measured in eggs were used to compute equivalent whole-body concentrations. The method is discussed in the text.
Values are arithmetic means and standard deviations.
(1)Not used in analysis, because tissue water content was unusually low (0.16) and lipid proportion was unusually high (0.18).
(2)Not used in analysis, because sample constituted prey remains, not whole body.

Estimation of concentrations in peregrine falcon prey. The gull database was used to estimate contaminant levels in gull prey of the peregrine falcon. Peregrine falcons can forage throughout the northern Channel Islands, and there is no information on the relative time spent on each island. Therefore, all western gull samples collected on Anacapa, Prince, San Miguel, Santa Cruz and Santa Rosa Islands (35 values in Table 5-4) were averaged together to estimate the average concentration in western gulls for use in the model. Concentrations for Heermann's and Bonaparte's gulls were set equal to those of California gulls, which were computed using 8 whole-body samples collected on Santa Cruz Island.

Concentrations of contaminants in whole bodies of Cassin's auklet were determined from egg measurements following the method discussed above for gulls. The value of 0.54 was used for the egg/whole body lipid-based ratio of contaminant levels. The egg/whole-body ratio of lipid contents was assumed to equal 1.0; this value was used for herring gull in the Great Lakes (Norstrom *et al*. 1986) and lies between two other estimates: 0.45 to 0.95 for American Kestrel (Wiemeyer *et al*. 1986) and 1.42 for western gulls on Santa Catalina Island. The values used to compute concentration were based on 16 samples collected on San Miguel, Santa Cruz and Prince Islands.

The water bird component of the diet was classified in six groups: grebes, shearwaters, waterfowl, shorebirds, phalaropes and "other alcids" (not including Cassin's auklet). The available data were converted to equivalent whole-body concentrations where necessary as described above. Data were available for grebes, shearwaters and "other alcids". The average contaminant levels for each of these groups were computed. Then, the overall average concentration for these three groups was applied to the remaining groups of waterbirds for which no data were collected (waterfowl, shorebirds and phalaropes). Finally, the computed concentrations for each group were weighted by their proportions in the peregrine diet (Hunt 1994, Appendix 5) and averaged. The overall average water bird contaminant levels were 1.19 ppm wet wb (p,p'DDE) and 0.39 ppm wet wb (total PCBs).

The land bird component of the diet is comprised of resident and migratory species. The concentrations of contaminants in the resident species was assumed to be zero. The concentrations of contaminants in the migratory species was estimated using published data on concentrations of p,p'DDE and total PCBs in migratory land birds sampled through the western U.S. and Mexico since 1980. Values were tabulated for species of land birds that have been observed on the Channel Islands (Jones *et al*. 1989). Concentrations measured in eggs were

converted to equivalent whole-body values using a conversion factor of 0.54. Muscle measurements were converted using a factor of 1.78.

These land bird values were then weighted by the proportional contribution of each species to the diet of the peregrine falcon, based on the information of Hunt (1994). Values were calculated for bird species that are migratory in the Southern California Bight, based on the Checklist of the Birds of Channel Islands National Park (Jones et al. 1989). A species was considered resident if it was recorded as resident on any of the Channel Islands (Jones et al. 1989). Sixty percent of the land bird diet was considered resident, and forty percent was considered migratory. The average concentrations of contaminants in whole bodies of migratory land birds of the Southern California Bight were estimated to be 0.33 p,p'DDE and 0.26 total PCB.

The dietary proportions, contaminant levels and whole-body composition information are given in Table 5-3.

5.2.5 Movement

During its first year, the peregrine falcon may move over large distances. Once the falcon reaches age two, it is considered resident on the island from which its eggs were sampled and indeed, Hunt (1994) found adults present in their aeries on the Channel Islands throughout the year. Data are available on contaminant levels in eggs collected on San Miguel, Santa Rosa, Santa Cruz, and Anacapa Islands; these populations are considered capable of feeding on bird prey sampled anywhere within those four islands. They are not considered likely to feed on prey captured on the mainland or on Santa Barbara, San Nicholas, Santa Catalina or San Clemente Islands.

5.3 DEVELOPMENT OF THE BALD EAGLE MODEL

5.3.1 Growth and Composition

Life history and growth rate. During the three months between hatching and fledging, bald eagle nestlings increase in weight from 100 grams to their adult weight (Stalmaster 1987). Average adult weights of bald eagles are approximately 5000 g (female) and 4000 g (male) (Stalmaster 1987). Bald eagles lay their first eggs in their fifth year of life (Stalmaster 1987).

Adult whole-body lipid content. Adult whole-body lipid content was estimated using the Patuxent Wildlife Research Center database, which includes measurements of bald eagles found dead throughout the United States over three decades (Patuxent Wildlife Research Center, U.S. Department of the Interior, Laurel, Maryland; see, for example, Wiemeyer 1984). These birds were collected in a variety of climates and died of a variety of causes. To estimate lipid contents for bald eagles from the Southern California Bight, an average lipid content was computed using data from eagles that (1) were collected in a region that is least likely to include migratory individuals and that is climatically similar to the Southern California Bight, and (2) died of causes not likely to result in emaciation.

Lipid contents measured in Florida bald eagles were used because of the mild climate with limited annual temperature variation and because of the existence of nonmigratory populations there (Stalmaster 1987). The distribution of lipid contents of eagles collected in Florida are presented in Figure 5-1. There appears to be a break in the distribution; in addition, birds with lipid contents greater than 0.03 glip/gwet fit a normal distribution (that is, a straight line on the probability plot), while birds with lipid contents less than 0.03 form a distinct subgroup and are not normally distributed. From this we summize that birds with less than 0.03 glip/gwet lipid may have been emaciated.

To eliminate emaciated birds, only those eagles with whole-body lipid contents greater than 0.03 were included in the calculation of average lipid content. In addition, only those eagles that died of electrocution, shooting, trauma, and hemorrhage were used, because these are the causes that are unlikely to result in emaciation. This resulted in a set of values that exhibited little seasonal variation (Figure 5-2). The average whole-body lipid contents was 0.051 (standard deviation 0.0109, range 0.037 - 0.067, n=13). This average was used in the model. Lipid content was not varied through the year, because the eagles of the Southern California Bight do not migrate, because the climate does not exhibit much seasonal variation, and because the Florida lipid data does not exhibit seasonal variation.

Adult whole-body dry weight. A value of 0.35 was used in the model, based upon the studies discussed in Section 5.2.

Egg lipid content. Lipid contents were measured in eggs of bald eagles collected in the Southern California Bight in 1992, as part of the Southern California Bight Damage Assessment. The average fresh weight-corrected value (+/- one standard deviation) was 0.036 +/- 0.064,
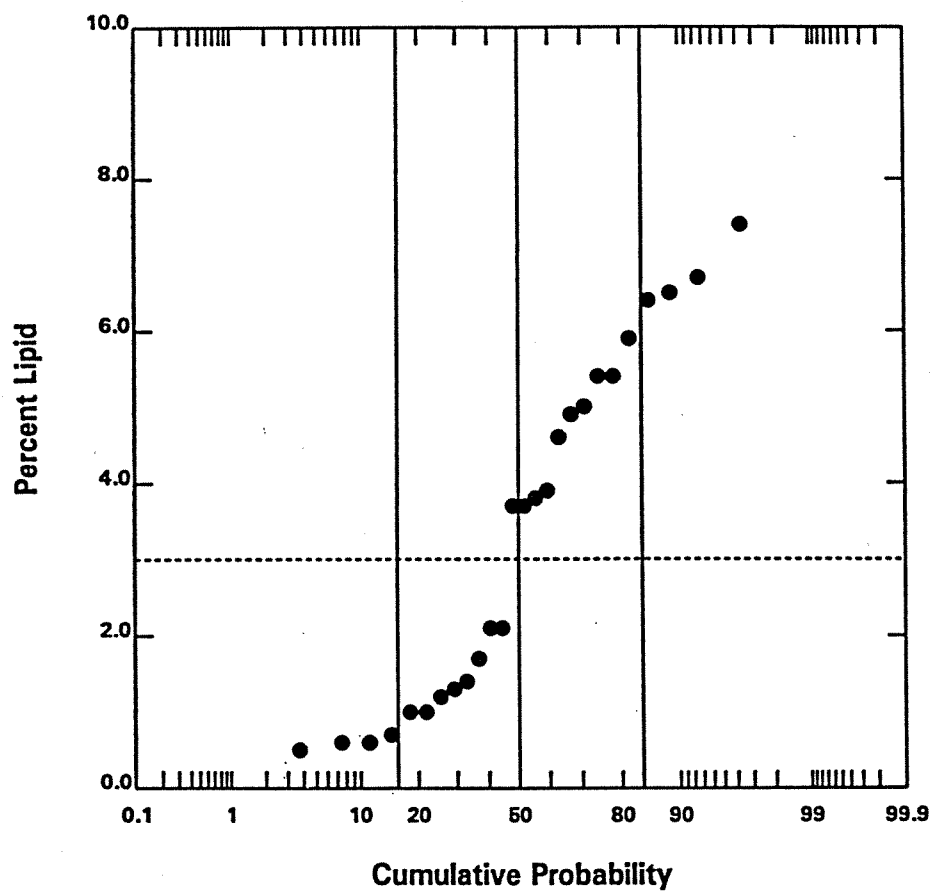
Figure 5-1.   Distribution of carcass lipid contents of adult bald eagles collected in Florida. Data of Paxtuxent Wildlife Research Center. All causes of death are included. The horizontal dashed line at 3% lipid indicates a break in the distribution; birds with lipid contents < 3% may be emaciated.
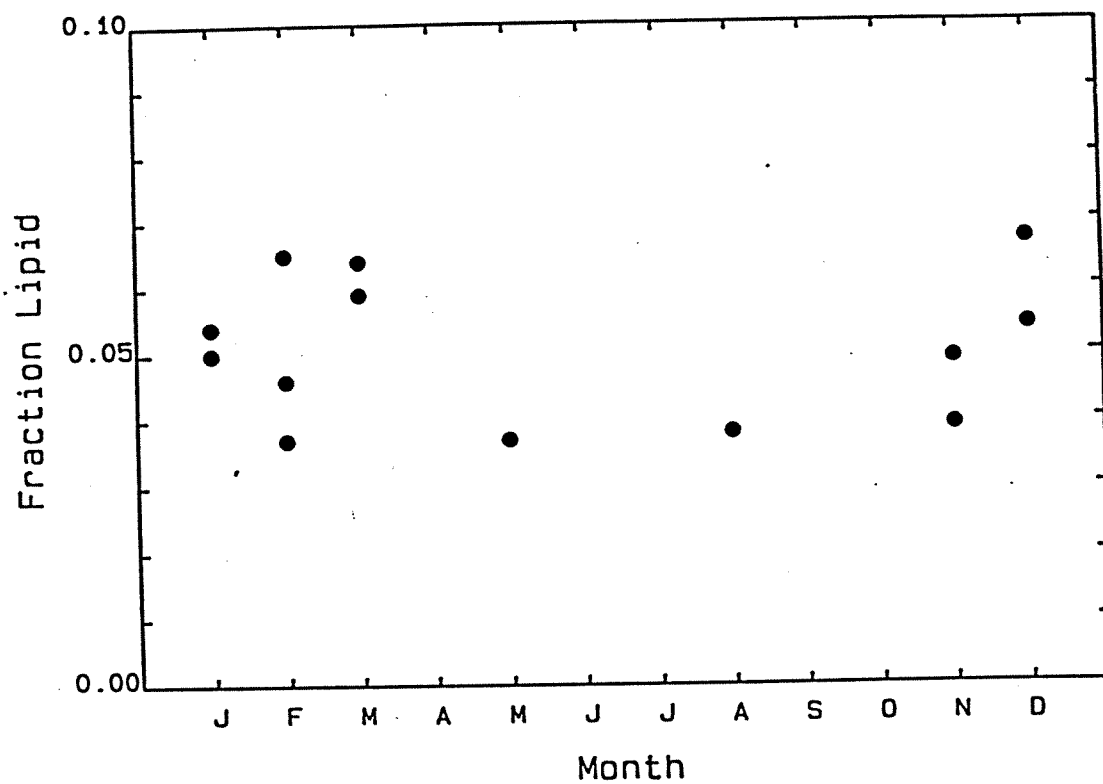
Figure 5-2.   Seasonal patterns in whole body lipid content for adult bald eagle carcasses collected in Florida. Data are from the Patuxent database. The causes of death are electrocution, shooting, trauma and hemorrhage.

n=9 (one value, 1.2, deleted). For comparison, Risebrough (1987) reported an average lipid content of 0.044 for bald eagle eggs (age unknown). As is the case for the peregrine falcon, it is likely that the value measured in the Southern California Bight underestimates the lipid content at the time of laying. Therefore, consistent with the treatment of the peregrine falcon, the literature-based value of 0.044 was used in the model.

The ratio of estimated average lipid contents between egg/whole body was 0.044/0.051 = 0.86 for the bald eagle. This value is within the range of values measured in captive American kestrels (average = 0.65, range = 0.45 to 0.95; Wiemeyer et al. 1986). Growth and body composition parameters for bald eagle are given in Table 5-5.

Table 5-5. Growth and Body Composition for the Bald Eagle

Stage	Time (days)	Weight (g(w))	Proportion Lipid
Yolk production[3]	(10 days)		
Egg laid	0 (early February)	99[1] (2/clutch)	0.044[2]
Egg hatched	37	90	0.03[1]
Nestling period	87	4200 (female) 3500 (male)	0.051[2]
Fledge	127	5000 (female) 4000 (male)	0.051[2]

NOTES: All values from Stalmaster 1987, EPA 1993, unless noted.
[1]Southern California Bight Damage Assessment data
[2]see discussion above
[3]Roudybush et al. 1979

5.3.2 Metabolic Rate

A metabolic rate of 0.68 kJ/gwet-d was computed for the bald eagle using Equation 5-1 (Williams et al. 1993). This value is similar to a total metabolic rate of 0.41 kJ/g-day for free-living bald eagles in Connecticut that was estimated using a time and energy budget (Craig et al. 1988, quoted by EPA 1993); the values are within 40 percent of each other. The value determined from the equation is used in the model because the data used to drive the equation are more robust and probably more accurate: it was developed using data from 62 species involving doubly-labeled water experiments, in which the metabolic rate is measured directly in free-living birds.

Measured and computed resting metabolic rates provide an additional independent check on the computed metabolic rate. A resting metabolic rate of 0.29 kJ/gwet-d was computed for a bald eagle using a model developed for the American kestrel and the white-tailed kite (Koplin et al. 1980, using a model developed by Kendeigh et al. 1977). This value is within a factor of 2 to 3 of the computed total metabolic rate, consistent with Walsberg (1983).

An average resting metabolic rate of 0.14 kJ/g-day was calculated from the measurements of 11 species of falconiformes in the thermoneutral zone performed by Wasser (1986). This value is more than a factor of 3 lower than the estimated total metabolic rate, which is consistent with the fact that the laboratory measurements were all collected in the thermoneutral zone, that is, the birds were not thermally stressed.

To provide another independent check of the computed metabolic rate, the food consumption rate was computed and compared with measured values. The food consumption rate for bald eagles calculated in the model ranged from 0.09 gwet/gwet-day (assuming that the prey was a bird with a fraction lipid of 0.10 and a fraction dry weight of 0.35, and that food assimilation efficiency was 0.8) to 0.12 gwet/gwet-day (assuming that the prey was a 50/50 mix of birds, along with fish with a fraction lipid of 0.052 and a fraction dry weight of 0.22, and that food assimilation efficiency was 0.8). These values are consistent with estimates for field populations that ranged from 0.065 to 0.12 for adult feeding on several food types (EPA 1993).

5.3.3 Feeding Habits

Food habits of the bald eagle were studied by David Garcelon et al. (1994a, 1994b) on Santa Catalina Island between December 1991 and July 1993. The opportunistic feeding habits of the bald eagle were demonstrated by seasonal variations in diet that correlated with availability of prey species. Prey species were identified from field observations of foraging eagles and at nesting sites.

Fish represented the greatest proportion of the eagle diet numerically, although birds and mammals were seasonally important. Over 28 species of fish were identified during the study. Contaminant concentrations have been measured in 9 of these species in the vicinity of Santa Catalina Island (Table 5-6). Sea lion carrion was the predominant mammal. Other mammalian prey included feral goats and unknown species. The birds were divided into four groups, based

on the dietary composition data of Garcelon *et al.* (1994a and 1994b): western gulls, other gulls, other waterbirds and land birds.

The proportions of each prey type in the diet throughout the year was estimated by Garcelon *et al.* (1994a and 1994b). Garcelon reported the total number of feeding observations for each species of prey over the entire period of observation, as well as the number of feeding observations for each month. The dietary composition based on total number of feeding observations was considered more accurate than the dietary composition given by month (Garcelon *et al.* 1994a and 1994b) and was used in the model simulations.

Table 5-6. Bald Eagle Fish Prey Species Near Santa Catalina Island in the HydroQual Database

Blacksmith	Mackerel
Kelp Bass	Opaleye
Black Surfperch	Barracuda
Garibaldi	Rockfish
Halfmoon	

On an energy basis, fish are still the dominant prey item (Table 5-7). Sea lions are more important on an energy basis than on a numerical basis. This is because sea lion prey is considered to be a mixture of muscle and blubber and has a high energy content (13.9 kJ/gwet, versus 8.0 - 8.8 for birds and 5.9 for fish). Invertebrates constituted on average 0.05 of the annual diet; the only identified invertebrate was squid. Data on contaminant levels in squid are not available; for purposes of the model invertebrates were considered to have the same contaminant levels as fish. The energy-based proportions were used to determine consumption rates for each prey type.

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Table 5-7. Dietary Composition, Contaminant levels and Body Composition for Prey of the Bald Eagle

	proportion of diet on a numerical basis	proportion of diet on an energy basis	contaminant levels (ppm wet weight)		body composition	
			p,p'DDE	PCB	fraction dry	fraction lipid
fish + invertebrates	0.86	0.79	0.11	0.07	0.24	0.03
sea lions	0.027	0.058	26.	5.2	0.40	0.30
other mammals	0.006	0.0073	0.0	0.0	0.35	0.06
birds western gulls	0.026	0.033	8.3	2.3	0.35	0.05
other gulls	0.009	0.013	5.4	1.3	0.38	0.06
waterbirds	0.062	0.085	1.7	0.60	0.38	0.06
land birds - resident	0.005	0.0078	0.0	0.0	0.38	0.06
land birds - migratory	0.004	0.0053	3.1	0.30	0.38	0.06
SUM	1.00	1.00				

Note: Arithmetic means used.
The proportion of the diet on an energy basis = proportion on a numerical basis X energy content, standardized to a total of 1.0.

5.3.4 Contaminant Levels in Prey

Bald eagle eggs were collected from Santa Catalina Island in 1992 as part of the Southern California Bight Damage Assessment. Contaminant levels for fish, birds and sea lions for Santa Catalina Island were calculated using data collected since 1985, as follows.

The HydroQual database includes four datasets for fish collected at Santa Catalina Island since 1985: Pollock (1991; muscle tissue), Garcelon (1994a and 1994b, whole bodies), other Southern California Bight Damage Assessment data (1992; liver tissue), and Risebrough (1987; muscle and liver tissue). The whole-body data collected by Garcelon (1994a and 1994b) were used directly. The data from Pollock (1991) included only total DDT; because p,p'DDE constitutes an average of 0.87 of total DDT in Southern California Bight fish tissues (average for all data in the HydroQual database), the Pollock values were multiplied by 0.87 before use.

The data collected by the other investigators were converted to a whole-body wet-weight basis as follows. First, whole-body concentrations were calculated by multiplying the lipid-based concentrations measured in these studies (ppm lipid) by the appropriate species-specific average lipid content (glip/gwet wb) computed from the data of Garcelon (1994a and 1994b). Next, species-specific average whole-body concentrations were calculated using the data from all four studies. Those species with less than four samples were combined to give one additional average. Finally, an overall average of the species-specific averages and the additional average

was computed. This overall average whole-body fish prey level was 0.11 ppm for p,p'DDE and 0.07 ppm for total PCBs.

As part of the Southern California Bight Damage Assessment, p,p'DDE and PCB concentrations were measured in sea lion tissues. Blubber samples were collected from San Miguel Island in 1991 (see Section 2.8). Remains of dead sea lions preyed upon by bald eagles were collected from Santa Catalina Island in 1992 (five samples) and 1993 (one sample). The remains included muscle, fat and skin of juveniles and adult females.

The contaminant dose received by a bald eagle preying on a sea lion carcass depends on the age of the sea lion and the tissue consumed by the eagle. The prey remains suggest that the eagles eat both fat and muscle from the carcass, although relative amounts are unknown. The age distribution of sea lion carcasses available to the bald eagles is unknown. In the absence of more specific data, we have assumed that the eagles consume tissue containing lipid-based contaminant concentrations equal to the average of the available data, including the San Miguel data and the Santa Catalina prey remains data. These averages are 86.8 μg p,p'DDE/g lipid and 17.4 μg PCB/g lipid (n=26). Because all body tissues have about the same lipid-based concentration, the contaminant concentration in the tissue eaten by the eagle is the product of the lipid-based concentration and the lipid content of the tissue. Based on a whole-body lipid content of 0.30 glip/gwet (see Section 4.3), the tissue concentrations are 26.0 ppm wet weight of p,p'DDE and 5.2 ppm wet weight of PCBs. The quantity of tissue eaten was calculated using 0.30 glip/gwet and 0.40 gdry/gwet weight.

Contaminant levels for bird prey on Santa Catalina Island were determined for each bird group mentioned above. For western gulls, all data values (whole body, egg and breast) were combined and averaged (a total of 38 samples). The means for Heermann's and California gulls were averaged and used to represent all other gulls (a total of 31 samples). For other waterbirds, all samples collected on islands in the Southern California Bight in each species named in the prey list were used to calculate an overall average for waterbirds (Table 5-8). For Cassin's auklets, Xantus' murrelet and Brandt's cormorants, the egg to whole-body conversions were performed as described above in the discussion of auklet levels for the peregrine falcon. The average water bird contaminant levels were 1.74 ppm whole body (p,p'DDE) and 0.60 ppm whole body (total PCBs).

All species of land birds in the contaminant concentration database were used with equal weighting, because information on species composition of the diet was insufficient to provide values for the proportional contributions of each prey species. The average concentrations of contaminants in whole bodies of migratory land birds of the Southern California Bight were estimated to be 3.13 (p,p'DDE) and 0.30 (total PCBs).

Table 5-8. Concentrations of Contaminants in Waterbird Prey of the Bald Eagle

Species	Location	Tissue	Year	No. Obs.	p,p'DDE	Total PCBs
Western grebe	S. Catalina	whoie body	1992	1	0.061	0.064
Sooty shearwater	S. Catalina	whole body	1992	1	1.920	0.062
Brandt's cormorant	S. Nicholas	egg	1992	15	1.63	0.78
Cassin's auklet	Prince Island	egg	1992	8	1.95	0.41
Cassin's auklet	S. Cruz	egg	1992	7	2.78	0.48
Cassin's auklet	S. Miguel	whole body	1993	1	0.36	0.075
Xantus' murrelet	S. Miguel	whole body	1992	1	0.98	0.38
Xantus' murrelet	S. Barbara	egg	1992	15	1.5	0.67

5.3.5 Movement

The bald eagles of Santa Catalina Island forage over a 3 to 5 km distance along the coast (Garcelon *et al.* 1994a and 1994b). In the model, they are considered to be resident (Garcelon *et al.* 1994a and 1994b) and to feed on fish, birds and sea lion carrion found near Santa Catalina Island.

5.4 DEVELOPMENT OF THE DOUBLE-CRESTED CORMORANT MODEL

5.4.1 Growth and Composition

Life history and growth rate. Double-crested cormorants reach near their adult weights within 35 days of hatching (Dunn 1975a). Average adult weight of double-crested cormorants from the Southern California Bight are 2,230 g (Ainley and Boekelheide 1990). They lay their first eggs in their fifth year of life.

Adult whole-body lipid content. An adult whole-body lipid proportion of 0.06 has been calculated for double-crested cormorants from New Hampshire (Dunn 1975). An average value of 0.07 was reported for double-crested cormorants in Green Bay, Wisconsin (range 0.01 to 0.12; Dale and Stromborg 1993). An annual average value of 0.07 glip/gwet was used in the model.

Adult whole-body dry weight.   A value of 0.37 was measured in double-crested cormorants from New Hampshire (Dunn 1975).   This value was used in the model.

Egg lipid content. Lipid contents were measured in eggs of double-crested cormorants collected in the Southern California Bight in 1992, as part of the Southern California Bight Damage Assessment. The average fresh weight-corrected value (+/- one standard deviation) was 0.038 +/- 0.006 (n=13). Gress *et al.* (1973) assumed a value of 0.042 for double-crested cormorant eggs in the Southern California Bight. This value was used in the model, because it is likely that the value measured in the Southern California Bight Damage Assessment underestimates the lipid content at the time of laying. For comparison, lipid contents for fresh double-crested cormorant eggs averaged 0.045 (collected in Green Bay, Wisconsin by Dale and Stromborg 1993) and 0.076 (collected in Lake Huron, Weseloh *et al.* 1983).

Growth and body composition parameters for the double-crested cormorant are given in Table 5-9.

Table 5-9. Growth and Body Composition for the Double-Crested Cormorant

Stage	Time (days)	Weight (g(w))	Proportion Lipid
Yolk production[5]	(10 days)		
Egg laid	0	41[1]	0.042[2]
	(May 15)[6]	(3.5/clutch)[6]	
Egg hatched	25[3]	32[3]	0.01[4]
	35	108[4]	0.02
	40	378	0.03
	45	759	0.04
	50	1222	0.05
	55	1705	0.06
	60	2032	0.063
Fledge	65	2300[6]	0.07
Reach final weight	92	2230[6]	0.07[2]

Notes:
[1]Southern California Bight Damage Assessment data.
[2]See text.
[3]Wiens & Scott 1975 - Brandt's Cormorant
[4]All proportion lipid data for nestlings from Dunn (1975). Nestling weights were based on Dunn (1975), adjusted approximately 10 percent to agree with hatchling and adult weights for the southern California Bight.
[5]Roudybush *et al.* 1979
[6]Ainley and Boekelheide 1990, Mitchell 1977, Vermeer 1969

5.4.2 Metabolic Rate

Daily energy expenditure of field populations of seabirds was measured using the doubly-labeled water technique (Figure 5-3; data reviewed by Ellis 1984). A relationship with body mass is evident, and is given below:

$$DEB = 6.9 * W^{-0.28} \qquad (5-3)$$

where DEB is in units of kJ/gwet-d and W is in units of g. This equation yielded a metabolic rate of 0.80 kJ/gwet-d for the double-crested cormorant. For comparison, resting metabolic rates measured in double-crested cormorants averaged 0.38 kJ/gwet-d (Henneman 1983). The value for total metabolic rate used in the model is 2.2 times this resting rate; this is consistent with the conclusion of Walsberg (1983) that total energy costs are 2 to 3 times basal.

An independent comparison with food consumption rates indicated that the model value is consistent with measured data. Food consumption rates for females calculated by the model were 0.20 gwet/gwet-d, assuming a food assimilation efficiency of 0.75 (Cummings 1987) and a lipid content of 0.052 and a dry weight of 0.22 in the fish prey. This value is within the range of 0.2 to 0.3 gwet/gwet-d estimated for free-living juveniles and adults of several species of cormorants (data reviewed by Dunn 1975b). In addition, Cummings (1987) estimated an annual average consumption rate of 0.26 gwet/gwet-d for double-crested cormorants in Florida, a population with a smaller body size and therefore probably with a higher metabolic rate.

5.4.3 Feeding Habits

In North America, double-crested cormorants feed on a variety of fish, with small amounts of crustacea and amphibians (Palmer 1962, quoted by Hunt et al. 1981). Samples obtained from material regurgitated by chicks on Prince Island in 1976 included "a common variety of mid-water fish that inhabit littoral waters particularly kelp beds" (Hunt et al. 1981). Species lists of prey of double-crested cormorants are given by Hunt et al. (1981) and Ainley et al. (1981).

$$DEB = 1.0W^{-0.28}$$
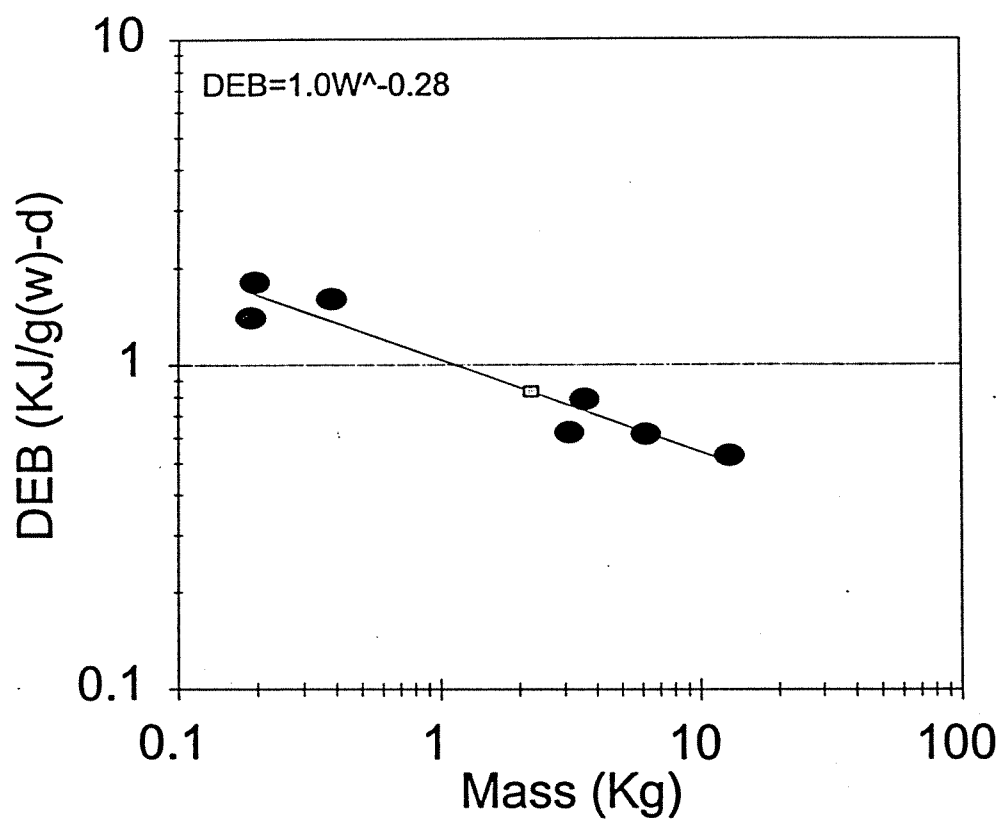
Source: Ellis 1984.

Figure 5-3. Daily energy budgets for seabirds as a function of body mass, based on double-labeled water studies. Original studies reviewed by Ellis (1984; ovals). The square represents the daily energy budget for double-crested cormorant from the Southern California Bight.

5.4.4 Contaminant Levels in Prey

Estimates of contaminant levels in prey of double-crested cormorants were developed from data on contaminant levels in fish samples contained in the HydroQual data base, using the species lists of Hunt *et al.* (1981) and Ainley *et al.* (1981) (Table 5-10).

Table 5-10. Double-Crested Cormorant Prey Species in HydroQual Database

Rockfish	Blacksmith
Sculpin	Vermillion Rockfish
Mackerel	Black Surfperch
Surfperch	

Prey levels were determined for a total of seven regions, as follows. The reason for using these regions is described below in the discussion of movement of the double-crested cormorants.

- within a 50 km radius of Anacapa Island
- within a 50 km radius of Santa Barbara Island
- Palos Verdes Shelf
- Santa Catalina, Santa Barbara, San Clemente Islands
- San Pedro Bay
- Santa Monica Bay
- northern Santa Monica Bay

To estimate a Bight-wide average fish prey concentration, the average of the last five regions in the above list was calculated.

Prey levels were estimated as for the bald eagle fish prey. Agencies represented in the HydroQual database for which data for double-crested cormorant prey exist include the Los Angeles County Sanitation District (LACSD), the Santa Monica Bay Restoration Study (SMBR), Risebrough (1987), Pollock (1991), and the Southern California Bight Damage Assessment. Data from all agencies were combined in calculating species averages. Because the tissue comparison performed for the bald eagle prey indicated no differences, all tissues were combined. Average lipid-based contaminant concentrations were computed for each species. An overall average prey concentration was computed by averaging the species-specific averages (ppm lipid). Next, the average whole-body lipid content for prey was estimated by averaging

the lipid contents of cormorant prey species collected on Santa Catalina Island by Garcelon *et al.* (1994a and 1994b; data for blacksmith, black surfperch and pacific mackerel). Finally, the overall average lipid-based concentration (ppm lipid) was multiplied by the average lipid content (glip/gwet wb) to yield an overall average whole body wet weight-based contaminant concentration (ppm wet wb) for prey of double-crested cormorants. Prey levels are presented in Table 5-11.

Table 5-11. Dietary Composition, Contaminant Levels and Body Composition for Prey of the Double-Crested Cormorant

	Contaminant Levels (ppm wet weight)			
	p,p'DDE	PCB	fraction dry	fraction lipid
Anacapa Island (50 km radius)	0.16	0.11	0.29	0.04
S. Barbara Is. (50 km radius)	0.13	0.05	0.29	0.04
Southern California Bight Regions:				
Palos Verdes Shelf	1.15	0.33	0.29	0.04
S.Barbara,S.Catalina,S.Clemente Is.	0.13	0.05	0.29	0.04
San Pedro Bay	0.35	0.35	0.29	0.04
Santa Monica Bay	0.33	0.18	0.29	0.04
Northern Santa Monica Bay	0.16	0.11	0.29	0.04
Average	0.42	0.20	0.29	0.04

5.4.5 Movement

Historically, the double-crested cormorant has inhabited several of the Channel Islands including Santa Catalina Island (Hunt *et al.* 1981). The double-crested cormorant populations on the California islands has been described as "mostly sedentary with populations moving ... from offshore islands to inshore channels in the nonbreeding season" (Hunt *et al.* 1981). The breeding season lasts from April through August (Hunt *et al.* 1981).

Double-crested cormorants tend to follow schools of fish, so their exact feeding locations are a function of the local oceanographic conditions that determine fish abundance (Ainley and Boekelheide 1990). They range widely to find food. Based on the work conducted in the Farallon Islands by Ainley and Boekelheide (1990) and observations in the Southern California Bight (Hunt *et al.* 1981), the following pattern of movement is used in the model. From fledgling to approximately 4 years of age, they can be found feeding anywhere within the Southern California Bight. Thereafter, during the breeding season, they are limited to a radius

of approximately 50 kilometers around the nest, a region that, for Anacapa and Santa Barbara Islands, does not include the Palos Verdes Shelf; during the rest of the year, they can feed throughout the Bight. The sensitivity of model results to this pattern is explored in a set of simulations in which each bird feeds in the vicinity of the breeding island throughout its lifetime.

Contaminant levels were measured in several fresh eggs collected in 1992 on Anacapa Island and in several rotten eggs collected on Santa Barbara Island. Model simulations were performed to compute egg levels in both locations. Prey levels were estimated for cormorants breeding on Anacapa Island and for birds breeding on Santa Barbara, and foraging within a radius of 50 kilometers around each island (adults in the breeding season); in addition, a Bight-wide average prey concentration was computed for young cormorants and for adults in the nonbreeding season (Table 5-11).

5.5 TOXICOKINETIC PARAMETERS

5.5.1 Gut transfer

In the fish and sea lion models, the ratio of assimilation efficiencies of contaminant/food is set at 0.75 for p,p'DDE and 1.0 for PCBs. Model results calculated here are reported using the same values. In a previous model of the bioaccumulation of several organic compounds in herring gulls (Clark *et al.* 1989), a ratio of 1.06 was used for all chemicals, including p,p'DDE and PCB congeners. Sensitivity analysis was performed for an assimilation efficiency of 1.06 for both chemicals.

5.5.2 Transfer to Eggs

Growth of the egg and contaminant transfer to it acts as a loss mechanism in the same way that body growth acts to dilute the body burden. In addition, estimating transfer to eggs is important, because levels in hatchlings are determined by levels in the egg. Finally, to compare model results with data requires an estimate of egg levels, because most data are for eggs.

The quantitative effect of egg production on female adult contaminant levels is determined by computing the amount of contaminant transferred to each egg, the number of eggs per clutch, and the number of clutches per year. Both dietary and body lipids are thought to contribute to

lipids in the egg (Norstrom *et al.* 1986). For example, Norstrom *et al.* (1986) found that lipid-based p,p'DDE levels in herring gull eggs were 0.40 times the whole-body levels, and concluded that some of the contaminant present in the egg originated in the food of the gull and some originated in body lipids. Dietary concentrations of highly bioaccumulated chemicals like p,p'DDE and PCBs are typically much less than the concentrations in the exposed birds; therefore, the lipid-based ratio of egg/body compartment contaminant is a direct estimate of the proportion of contaminant that comes from body lipid. For example, an egg/body compartment value of 0.5 indicates that the egg contains approximately 50 percent body lipids and 50 percent dietary lipids; a value of 1.0 indicates that approximately 100 percent of egg lipids originate in body lipids.

As the embryo develops, the lipid content declines, because the embryo utilizes the lipid as an energy source. For example, more than 1/3 of the fat in a hen's egg is depleted during development of the chick (Stickel *et al.* 1973, quoting Romanoff 1932). Alaskan peregrine falcon eggs contained 0.044 glip/gwet, whereas downy chicks contained 0.033 glip/gwet (Cade *et al.* 1968). The lipid content of herring gull eggs declined by a factor of 2 during incubation (Peakall and Gilman 1979). Thus, the lipid-based concentration of contaminants in the egg increases with the age of the egg, because the lipid content decreases while the total amount of contaminant does not change. Estimates of lipid-based egg/body contaminant ratios must include a consideration of the age of the egg.

Ratios of egg/body compartment lipid-based levels of p,p'DDE and PCBs are presented in Table 5-12. Values range from 0.31 to 0.74 (not including two high values because of uncertainties in the data)[6]. In approximately half of the cases, the eggs were "fresh," which includes eggs collected "during the laying period;" presumably this involves a period of a few days. The differences between fresh and unknown eggs and between PCBs and DDE were not significant (Student's t-tests, P=0.05). Therefore, all data were combined to yield an overall lipid-based egg/body ratio of 0.54 (standard deviation = 0.13, n=17).

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[6] The second-highest value, 0.96 for the sparrowhawk, was measured on a single dead adult and its eggs collected in the field; the birds in this study had been found dead and therefore may have been starved or suffering toxic effects of contaminants (Bogan and Newton 1977). The highest value in Table 5-13 (egg/muscle = 1.0) is somewhat uncertain, because it was based on a series of relationships (egg/plasma, plasma/brain, brain/muscle) measured in several species in two studies.)

Table 5-12. Egg/Body Compartment Contaminant Ratios, Lipid Basis

Species	Compartments	Chemical	Ratio of Contaminants	Age of Egg	Ref.
Peregrine falcon	egg/fat	DDE	0.39	unknown	Cade et al. 1968
Peregrine falcon	egg/muscle	DDE	0.58	unknown	Cade et al. 1968
American Kestrel	egg/carcass	DDE	0.54	variable	Wiemeyer et al. 1986
Sparrowhawk	egg/whole body	DDE	0.96	unknown; birds found dead	Bogan & Newton 1977
Birds of prey	egg/muscle[2]	DDE	1.00	variable	Henny and Meeker 1981
California gulls	egg/abdominal	DDE	0.48	fresh	Vermeer and Reynolds 1970
Herring gull	egg/whole body	DDE	0.40	fresh[1]	Norstrom et al. 1986
Herring gull	egg/whole body	DDE	0.56	unknown	Braune and Norstrom 1989
Herring gull	egg/muscle	total DDT	0.64	unknown	Lemmetyinen et al. 1982
Arctic tern	egg/muscle	total DDT	0.31	unknown	Lemmetyinen et al. 1982
Adelie penguin	egg/abdominal fat	DDE	0.51	fresh	Tanabe et al. 1986
Adelie penguin	egg/sub-cutaneous fat	DDE	0.70	fresh	Tanabe et al. 1986
Double-crested Cormorant	egg/carcass	DDE	0.49	fresh	Dale and Stromborg 1993
Hen	egg/body fat	DDE	0.72	presumed fresh	Kan and Rooyen 1978
Double-crested Cormorant	egg/carcass	PCB	0.74	fresh	Dale and Stromborg 1993
Herring gulls	egg/whole body	PCB congeners	0.47	unknown	Braune and Norstrom 1989
Herring gulls	egg/muscle	PCB	0.50	unknown	Lemmetyinen et al. 1982
California gulls	egg/abdominal	PCB	0.41	fresh	Vermeer and Reynolds 1970
Arctic tern	egg/muscle	PCB	0.69	unknown	Lemmetyinen et al. 1982

NOTES: (1)"fresh" includes eggs collected "during the laying period"
(2)This value was based on regressions between plasma and egg contents for several birds of prey (Henny and Meeker 1981), between plasma and brain for several birds of prey (Henny and Meeker 1981) and between brain and muscle in sparrow hawks (Bogan and Newton 1977).

Thus, dietary lipids are assumed to contribute an average of 1.00-0.54=0.46 of egg lipids, with a range of 0.26 to 0.69. In the model calculations presented below, the average value is used. The sensitivity of model results to uncertainty in the value of this parameter is discussed below.

5.5.3 Excretion

Whole-body excretion rates of p,p'DDE and PCBs have been measured in birds (Table 5-13). The p,p'DDE rates are plotted in relation to body weight in Figure 5-4. A regression through the data resulted in a slope of -0.30, the same value as the slope of log(overall metabolic rate) vs. log (body weight) (Equation 5-2)[7]. This relationship is consistent with metabolic scaling of many physiological processes that exhibit weight-based relationships with powers on the order of -0.25 to -0.35 (Peters 1983). It is also consistent with the weight-based relationship observed in the PCB excretion rate data for mammals discussed in Section 4.

These data probably overestimate the excretion rates for the species of interest. This conclusion is based upon the following: First, the elimination of lipophilic pollutants by terrestrial animals appears to be controlled by enzymatic modification of the contaminants (Walker 1987). "For many liposoluble pollutants, which are not esters (e.g., dieldrin, DDE, PCBs), hepatic microsomal mono-oxygenase (HMO) is the most effective (often the only) enzyme which can attack them, albeit very slowly" (Walker 1987, page 235). For example, the rate of excretion of unchanged p,p'DDE is very slow; the measured half-life of unchanged p,p'DDE from the feral pigeon (Columba livia) was approximately 11 years (Sidra and Walker 1980; Walker 1990). However, p,p'DDE disappears from the pigeon relatively rapidly; the overall measured half-life of p,p'DDE in the feral pigeon was approximately 250 days (Bailey et al. 1969). These results suggest that the overall rate of loss of p,p'DDE is determined not by excretion of unchanged p,p'DDE, but primarily by its rate of metabolism. In support of this, DDE and PCBs are known inducers of HMO enzymes (Ronis and Walker 1989). Finally, HMO activity shows a relationship to body weight that is similar to that observed for whole-body excretion rate. Ronis and Walker (1989) computed weight-based relationships with slopes approximately equal to -0.3.

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[7] The regression line presented in Figure 5-4 excluded one outlying point (cackling goose); Clark et al. 1987 suggest that this value is high because of the increased circulating lipid levels that may occur during this species' migration.

0.1

0.01

goose(f)

quail(c)

grackle(c)

pigeon(c)

gull(f)

chicken(c)

penguin(f)

0.001

0.0001

0.01   0.1   1   10

Body Weight (Kg wet)

Excretion rate (1/d)

Log(excretion rate in 1/days) = -2.7 - 0.30 X log(weight in kg)
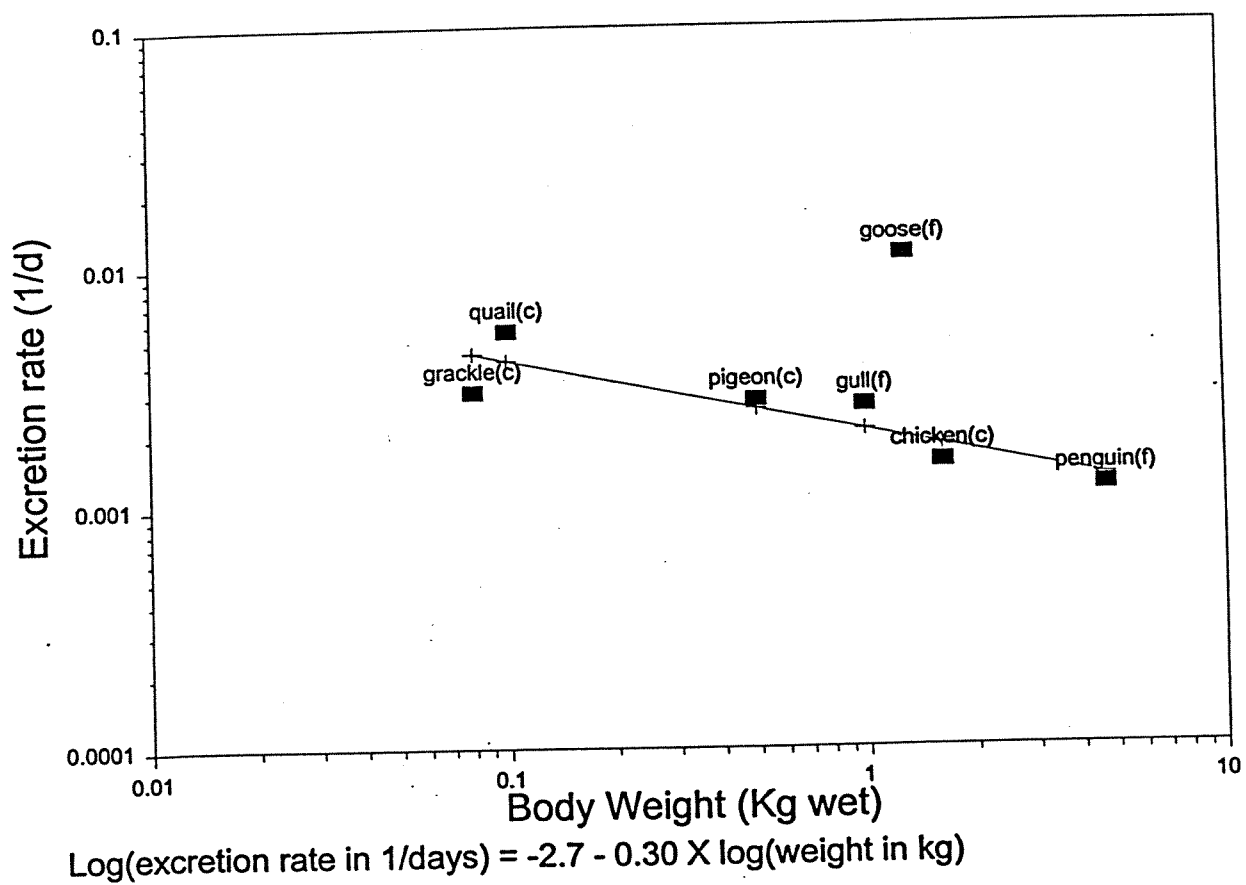
Figure 5-4. p,p'DDE excretion rates in birds as a function of body weight. Values based on literature review (see Table 5-13). Studies performed on caged birds are indicated by (c); field studies indicated by (f). Line represents best fit regression.

Table 5-13. Elimination Rates in Birds

Species	Weight (Kg)	Reference	Experiment Duration	Field/C age	Half-life (days)	Reference
			DDE			
Grackle	0.08	1	112 da	cage	229	2
Pigeon	0.5	3	288da	cage	250	4
Herring gull	1.0	5	1 yr	field	264	5
Adelie penguin	4.8	11	5 yr	field	580	9
Cackling goose	1.3	6	276 da	field	63	6
Japanese Quail	0.1 (4 wks old)	8	70 da	cage	128	8
Shaver starcross 288 high-producing hens (p,p'DDT)	1.6	12	84 da	cage	450	12
			PCBs			
Grackle (Aroclor 1254)	0.08	1	224 da	cage	89	7
Puffins	0.6	11	3 yr	field	488	10
Adelie penguin	4.8	11	5 yr	field	270	9

References and notes:
1 Spector 1956
2 Stickel *et al.* 1984a
3 Newell *et al.* 1987
4 Bailey *et al.* 1969
5 Norstrom *et al.* 1986. The value of 284 days was estimated by averaging the corrected wet weight-based excretion rates for whole body, liver, brain and muscle.
6 Anderson *et al.* 1984
7 Stickel *et al.* 1984b
8 Fawcett *et al.* 1981

9 Subramanian *et al.* 1987
10 Harris & Osborn 1981, quoted by Walker 1992
11 Dunning 1993
12 Kan and Rooyen 1978

Second, measured rates of HMO activity are greater in omnivorous and herbivorous species than in piscivorous and carnivorous species (Table 5-14; Ronis and Walker 1989). Thus, the predators are likely to eliminate these contaminants slower than the omnivores and herbivores. Finally, the excretion rates were measured in omnivorous and herbivorous bird species, while the species of interest are predatory. Five of the six values in the regression in Figure 5-4 were measured in herbivorous or omnivorous species (grackle, pigeon, herring gull,

Japanese quail and domestic fowl).[8] In summary, it is likely that the excretion rates reported in Table 5-13 and Figure 5-4 are greater than the rates in the three species of interest.

Table 5-14. Relative HMO Activities in Birds

Species		Relative HMO Activity[1]
Herbivorous or omnivorous species with measured p,p'DDE excretion rates		
Japanese quail	Coturnix japonica	0.74
Domestic fowl	Gallus domesticus	0.64
Pigeon	Columba livia	0.12
Herring gull	Larus argentatus	1.07
Species of interest or their relatives		
Double-crested cormorant	Phalacrocorax auritus	0.042
Sparrowhawk	Accipiter nisus	0.057
Buzzard	Buteo buteo	0.074
Kestrel	Falco tinnunculus	0.23

Note: [1]Relative activity defined relative to activity in the male rat. All data taken from Ronis and Walker (1989), who reviewed values measured in several studies by several investigators.

Therefore, the measured excretion rates were reduced in order to provide more realistic values for the three predatory species of interest. Excretion rates for all three species and for both chemicals were estimated in a consistent fashion, subject to several constraints. The accuracy and realism of the model results is improved to the degree that the resulting excretion rates satisfy these constraints:

(1) The rates for the species of interest should be lower than the measured rates (Figure 5-4 and Table 5-13). The difference is likely to be less than a factor of 10, based on the HMO activities in Table 5-14.

(2) The excretion rates chosen for all three species should exhibit a weight-based relationship with a slope of approximately -0.3, consistent with measured metabolic rates, the measured excretion rates and general biological principles. This should be true for p,p'DDE and total PCBs.

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[8]The only exception is the Adelie penguin (Pygoscelis adeliae), which is a piscivorous species. This rate was determined by calculation, and not by direct measurement; it was estimated for the period between birth and five years of age using measured contaminant loads in five-year-olds along with estimates of food consumption rates and prey concentrations for the first five years of life. Thus, the uncertainty associated with the penguin calculation is greater than for the other species.

(3)   The PCB excretion rates should be approximately 1 to 3 times the p,p'DDE excretion rates. This is based on several estimates of the relationship between PCB and p,p'DDE excretion rates in birds. The measured PCB excretion rate for the grackle was 2.6 times the measured p,p'DDE rate (Stickel *et al.* 1984a and 1984b); a ratio of 2.2 was estimated for the adelie penguin based on field data (Subramanian *et al.* 1987). An additional independent estimate of the ratio of excretion rates was based upon the observation that the measured bioaccumulation factor for PCBs in herring gulls was within 10 percent of the bioaccumulation factor for p,p'DDE (Braune and Norstrom 1989). For these to be equal, the excretion rates must have a ratio approximately equal to that of the contaminant assimilation efficiencies. In the present model, the ratio of PCB/p,p'DDE assimilation efficiencies is 1.0/0.75 = 1.33 (see above), implying that the ratio of excretion rates should be approximately equal to 1.3. Thus, the ratio of the computed PCB/p,p'DDE excretion rates should be consistent with the three data-based estimates (1.3, 2.2, 2.6).

(4)   The models should compute trophic transfer factors (ratios of predator egg/prey contaminant levels) that are consistent with field-measured egg/prey ratios.

(5)   The models should compute similar egg/prey ratios for PCBs and p,p'DDE (Braune and Norstrom 1989)

Egg/prey contaminant ratios. A literature review was conducted to find laboratory and field estimates of the ratios of p,p'DDE and PCB levels in bird- and fish-eating birds and in their prey (Tables 5-15 and 5-16). Predator-prey ratios were either given by the authors, or were estimated from their data on contamination levels and on species composition of the diet. Values were reported on a wet-weight basis for both predator and prey.

The computed egg/prey ratios should be similar among predatory species. The steady-state egg/prey ratio is proportional to the ratio of dose/loss rates. The dose received by the birds is proportional to the food consumption rate, which is proportional to the metabolic rate. The rate of loss by the birds is approximately proportional to the excretion rate (egg production has a relatively minor impact). Therefore, the egg/prey ratio is approximately proportional to the ratio of metabolic rate/excretion rate. Because both of these parameters are approximately proportional to weight$^{-0.3}$, changes in weight make little difference to the egg/prey ratio.

Table 5-15. Ratios of DDE Levels in the Eggs of Predatory and Piscivorous Birds and Their Prey

Species	Weight (kg)	Prey	Source of Birds (study duration)	Movement	Ratio Between Compartments	Age of Egg	Reference
Peregrine Falcon	1.0	Birds	Yukon River, AK	migratory	16	unknown	Cade et al. 1968
Peregrine Falcon	1.0	Birds	Rocky Mountains, CO, AZ and NM	at least weakly migratory	23	addled eggs; variable age	Enderson et al. 1982
Peregrine Falcon	1.0	Birds	Rocky Mountains, CO, AZ and NM	migratory	6.8	addled eggs; reported ppm dry with percent water factor	Ellis et al. 1989
Bald Eagle	4.5	Birds	Klamath Basin, OR	resident	47	variable - corrected to fresh	Frenzel 1985
Double-crested Cormorant	2.1	Fish	Lake Huron	migratory	57	fresh	Weseloh et al. 1983

Notes: Wet weight basis
w.b. = whole body

Table 5-16. Ratios of PCB Levels in the Eggs of Predatory and Piscivorous Birds and Their Prey

Species	Weight (kg)	Prey	Source of Birds (study duration)	Movement	Ratio Between Comparments	Age of Eggs	Reference
Double-crested Cormorant	2.1	Fish	Lake Huron	migratory	45	fresh	Weseloh et al. 1983
Double-crested Cormorant	2.1	Fish	Lake Poinsett and Dry Lake, S. Dakota	migratory (?)	75	unknown	Greichus et al. 1973
Double-crested Cormorant	2.1	Fish	S. Green Bay, Wisconsin	migratory	18	unknown[1]	Dale & Stromborg, 1993[2]
Double-crested Cormorant	2.1	Fish	N. Green Bay, Wisconsin	migratory	9.1	half to fully incubated	Yamashita et al. 1993[2]
Black-Crowned Night Heron	-	Fish	S. Green Bay, Wisconsin	?	3.5	unknown[1]	Dale & Stromborg 1993[2]
White Pelican	-	Fish	Lake Poinsett and Dry Lake, S. Dakota	migratory	22	unknown	Greichus et al. 1973
Forster's Tern	-	Fish	S. Green Bay, Wisconsin	?	4.4	unknown	Ankley et al. 1993[2]
Forster's Tern	-	Fish	S. Green Bay, Wisconsin	?	4.1	unknown[1]	Dale & Stromborg 1993[2]
Common Tern	-	Fish	S. Green Bay, Wisconsin	?	6.2	unknown[1]	Dale & Stromborg 1993[2]
Caspian Tern	~1.0	Fish	N. Green Bay, Wisconsin	?	14	unknown	Yamashita et al 1993[2]

Notes: Wet weight basis
(1) Corrected for mositure loss
(2) Fish levels for Green Bay from HydroQual (1995), Connolly, et al. (1992)

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Therefore, the egg/prey ratios should differ among species primarily insofar as there are species-specific differences in the ability to metabolize these chemicals that are not approximately proportional to weight$^{-0.3}$. The observation of the negative relationships between weight and excretion rate in omnivores and herbivores as well as between weight and HMO activity suggests that this is unlikely.

p,p'DDE values for field populations of raptors with avian prey averaged 23 (range: 6.8 to 47, n=4). One additional value was measured for a double-crested cormorant (57). Values for piscivorous species are expected to be greater than values for bird-eating species, because the energy content of fish is generally 0.5 to 0.7 times that of birds; this difference in energy content leads to a relatively greater consumption rate in order to meet the energy requirements of the cormorants. When the cormorant value is adjusted to account for the difference in energy content, the resulting value is consistent with the other four (57 * 6./8.3 = 41).

PCB values for field populations of piscivores averaged 20 (range 3.5 to 75; n=10). The similarity between the average p,p'DDE and total PCB egg/prey ratios is consistent with our understanding of toxicokinetic mechanisms (see above).

Estimation of excretion rates. Excretion rates for all three species and both chemicals were determined by calibrating the models to the measured egg/prey contaminant ratios with the constraints described above. The distributions of measured egg/prey ratios are presented in Figure 5-5 (symbols). The top panel presents egg/prey ratios for p,p'DDE in bird-eating species[9]. The bottom panel presents egg/prey ratios for total PCBs in piscivorous species. Model excretion rates were adjusted until the computed egg/prey ratios were close to the geometric means of the data for both chemicals, satisfying constraint (4). The horizontal lines in Figure 5-5 represent the resulting egg/prey ratios for the three species of interest.

The excretion rates that resulted in the egg/prey ratios shown in Figure 5-5 are presented in Table 5-17 as half-lives. The p,p'DDE excretion rates, corrected for differences in lipid content[10], are plotted in Figure 5-6, along with the data for the omnivores and herbivores. The

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[9]Note that in the top panel of Figure 5-6, the adjusted cormorant value is included.

[10]The correction for differences in lipid content involved multiplying the excretion rate by the ratio of whole-body fraction lipid of the species/ 0.10. This was done to make the excretion rates comparable amongst themselves and with the data.
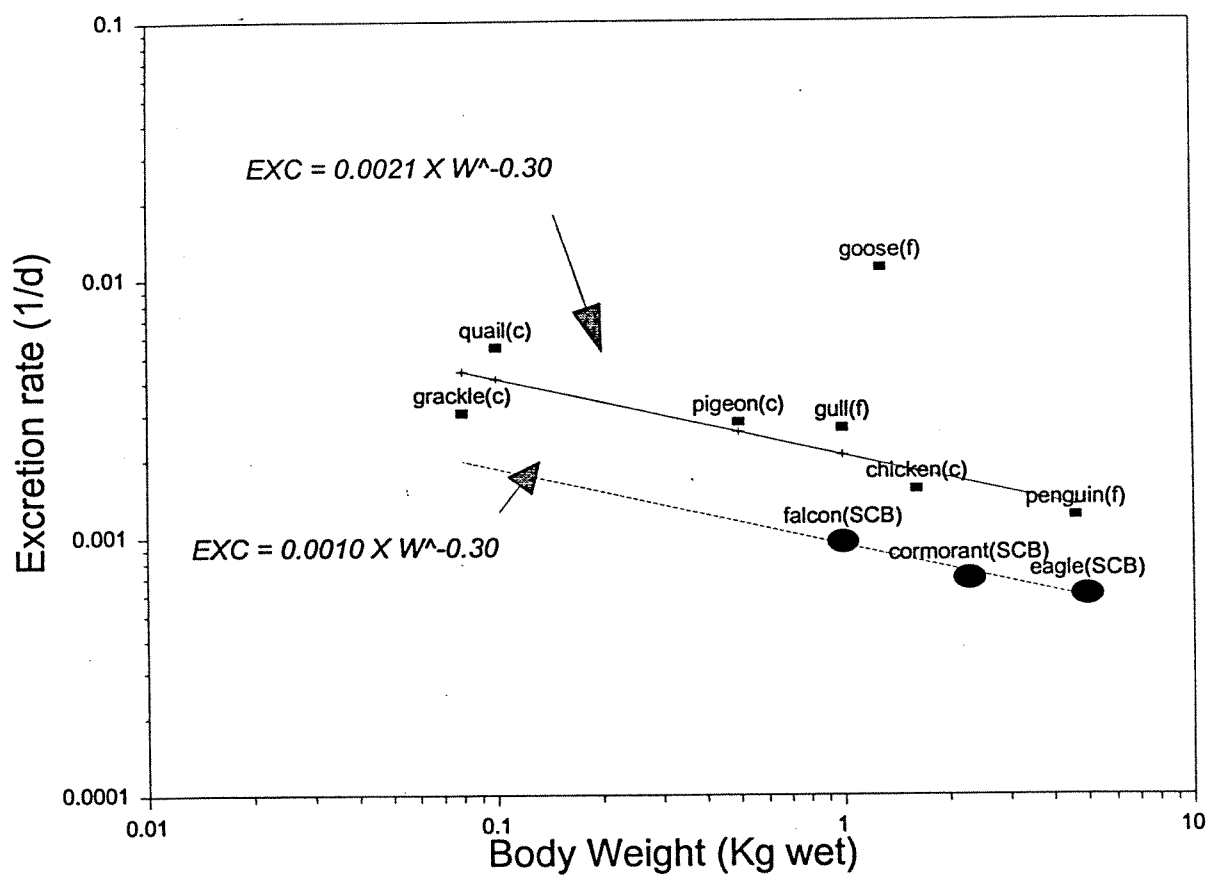
Figure 5-5. p,p'DDE excretion rates in birds - data for omnivorous and herbivorous species (squares) and computed values for predators (ovals). Studies performed on caged birds are indicated by (c); field studies indicated by (f). Lines represent best fit regressions.

ppDDE Egg/Prey Ratio with Bird Prey

PCB Egg/Prey Ratio with Fish Prey
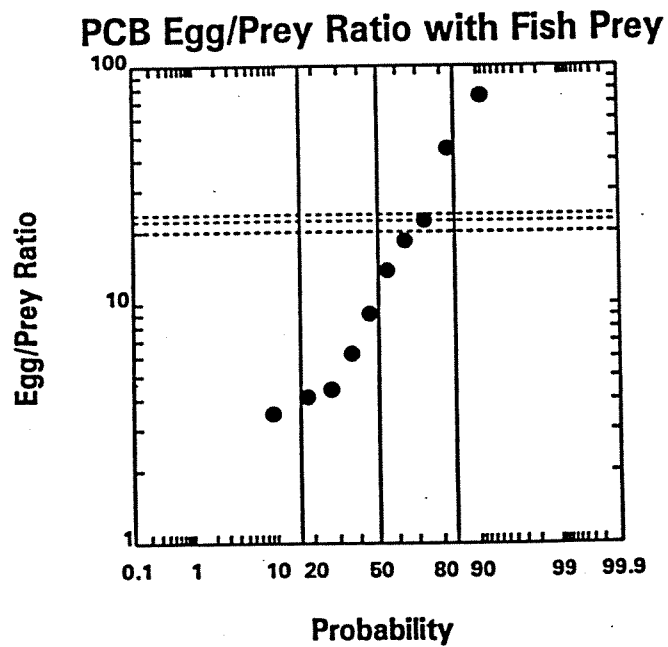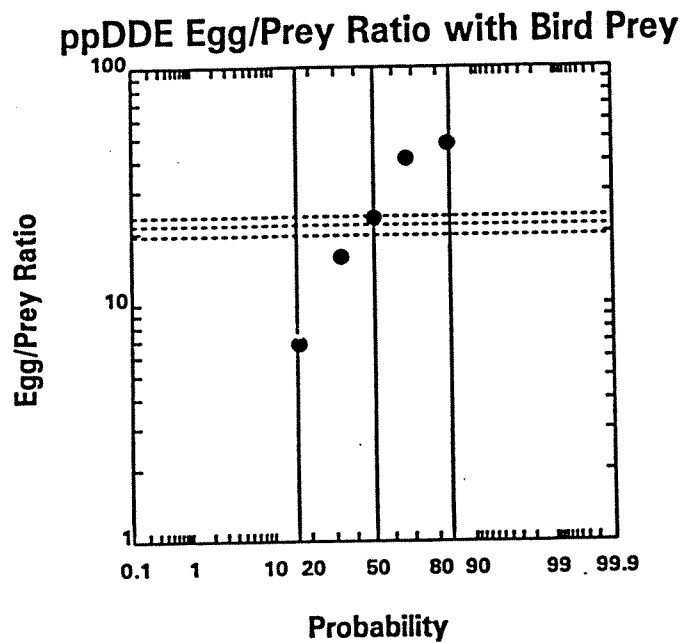
Dashed lines: Model computations for three species

Figure 5-6.   Egg/prey ratios in birds - data and model computation. Symbols represent data in Tables 5-15 and 5-16.

excretion rates for the species of interest are lower than for the omnivores and herbivores by a factor of two, satisfying constraint (1). The three rates are proportional to weight $^{-0.30}$, satisfying constraint (2). The PCB excretion rates are two times greater than the p,p'DDE rates, consistent with measurements (Table 5-17); this satisfies constraint (3). Finally, the computed egg/prey ratios for PCBs lie within 50 percent of the p,p'DDE ratios when corrected for differences in prey energy content, satisfying constraint (5) (Figure 5-5).

Table 5-17. Excretion Half-Lives for p,p'DDE and Total PCBs in Model Birds

Species	p,p'DDE	Total PCBs	DDE/PCB
Peregrine falcon	544	272	2.0
Bald eagle	594	297	2.0
Double-crested cormorant	705	352	2.0

Notes:
Values in units of days.

One uncertainty in the estimation procedure for excretion rates is the whole-body lipid content of the adult birds. The egg/prey ratios that were used to estimate excretion rate were measured on a wet-weight basis (ppm wet egg / ppm wet prey). To assess the degree to which variation in whole-body lipid content might affect the wet weight-based egg/prey ratios, a series of simulations were performed using the peregrine falcon model. Simulation results indicate that variation in whole-body lipid content has a relatively small impact on levels in the eggs (Figure 5-7, bottom panel). That is, the computed wet weight-based egg/prey ratios are only minimally affected by variation in the female whole-body lipid content. This is because the transfer to the eggs is determined by the lipid-based contaminant concentration in the female which is only minimally affected by changes in the lipid content. In contrast, variation in the average female whole-body lipid content has a direct impact on computed wet weight-based contaminant levels in the female (Figure 5-7, top panel). This is because when female lipid content doubles, excretion is halved, so whole-body contaminant level increases approximately two-fold.

In conclusion, the excretion rates for all three species for both chemicals were computed in a consistent fashion. The consistency among all of the values based on several independent lines of evidence has resulted in a set of models that can be used to provide a realistic analysis of bioaccumulation in the Southern California Bight.
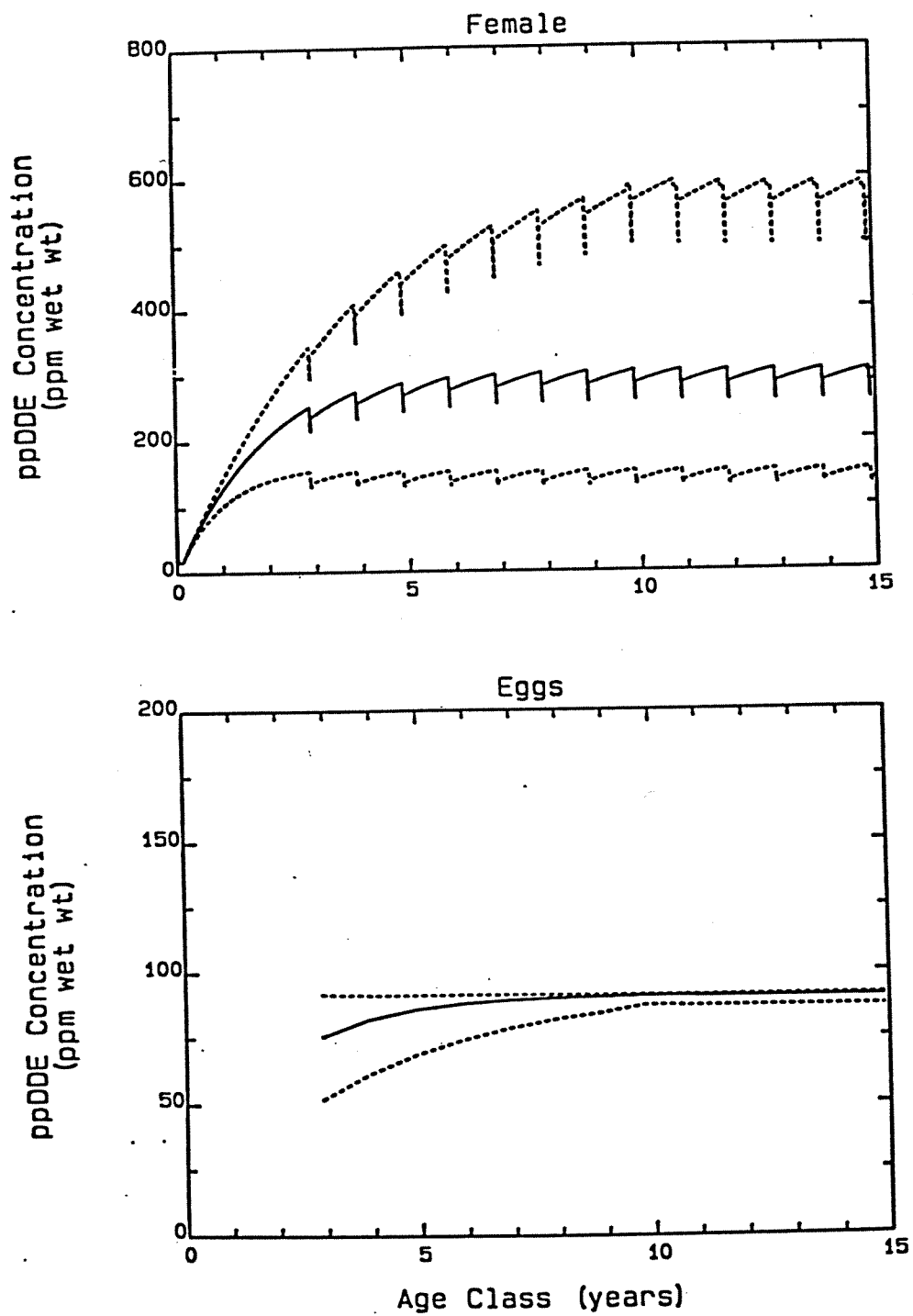
Figure 5-7. Computed wet weight p,p'DDE concentrations in female peregrine falcons and their eggs. Female lipid fraction = 0.075 (solid line); female lipid fraction = 0.0375 and 0.15 (dashed lines).

5.6   RESULTS OF SAMPLE STEADY-STATE MODEL SIMULATIONS

To explore contaminant dynamics over the lifetime of the birds, model simulations were performed in which prey contaminant levels were kept constant (Figures 5-8, 5-9, and 5-10). Concentrations decline immediately following hatching due to growth dilution. Thereafter, concentrations rise. Following the start of reproduction, contaminants are lost to the eggs each year. Egg loss causes the adjustment in whole-body contaminant levels seen annually in adult birds in Figures 5-8, 5-9 and 5-10.

By age 5, soon after the start of reproduction, the p,p'DDE level in peregrine falcons reach steady state (Figure 5-8). Steady state is achieved later in the bald eagle (Figure 5-9) and the double-crested cormorant (Figure 5-10). This difference is due to differences in excretion rate: fastest for the peregrine falcon, slower for the bald eagle and for the double-crested cormorant.

5.7   SENSITIVITY OF THE MODEL TO PARAMETER UNCERTAINTY

Fraction of dietary lipids in the egg. Uncertainty in the fraction of dietary lipids in the egg has a direct impact on computed contaminant levels in the egg and a relatively small impact on levels in the female. The concentrations of p,p'DDE in peregrine falcon females and their eggs are presented in Figure 5-11. The solid lines represent model simulations in which the fraction of dietary lipid in the egg = 0.46; the dashed lines represent fractions of 0.26 and 0.69, the range of fraction of dietary lipid in the egg, computed from the range of egg/whole body contaminant ratios in Table 5-12. The effects of this uncertainty on computed egg levels in the Southern California Bight are presented below.

Contaminant assimilation efficiency. Changes in the contaminant assimilation efficiency results in proportional changes in contaminant levels in both the female and the egg. Thus, if the assimilation efficiency of p,p'DDE relative to food is changed from 0.75 to 1.06, contaminant levels increase 41 percent. If the assimilation efficiency of total PCBs is changed from 1.0 to 1.06, contaminant levels increase 6 percent.

Sex-dependent patterns in contaminant levels. The development and laying of eggs causes changes in the contaminant levels in the female: during the ten-day period of egg formation, the contaminant level drops due to growth dilution. In the model, the eggs are
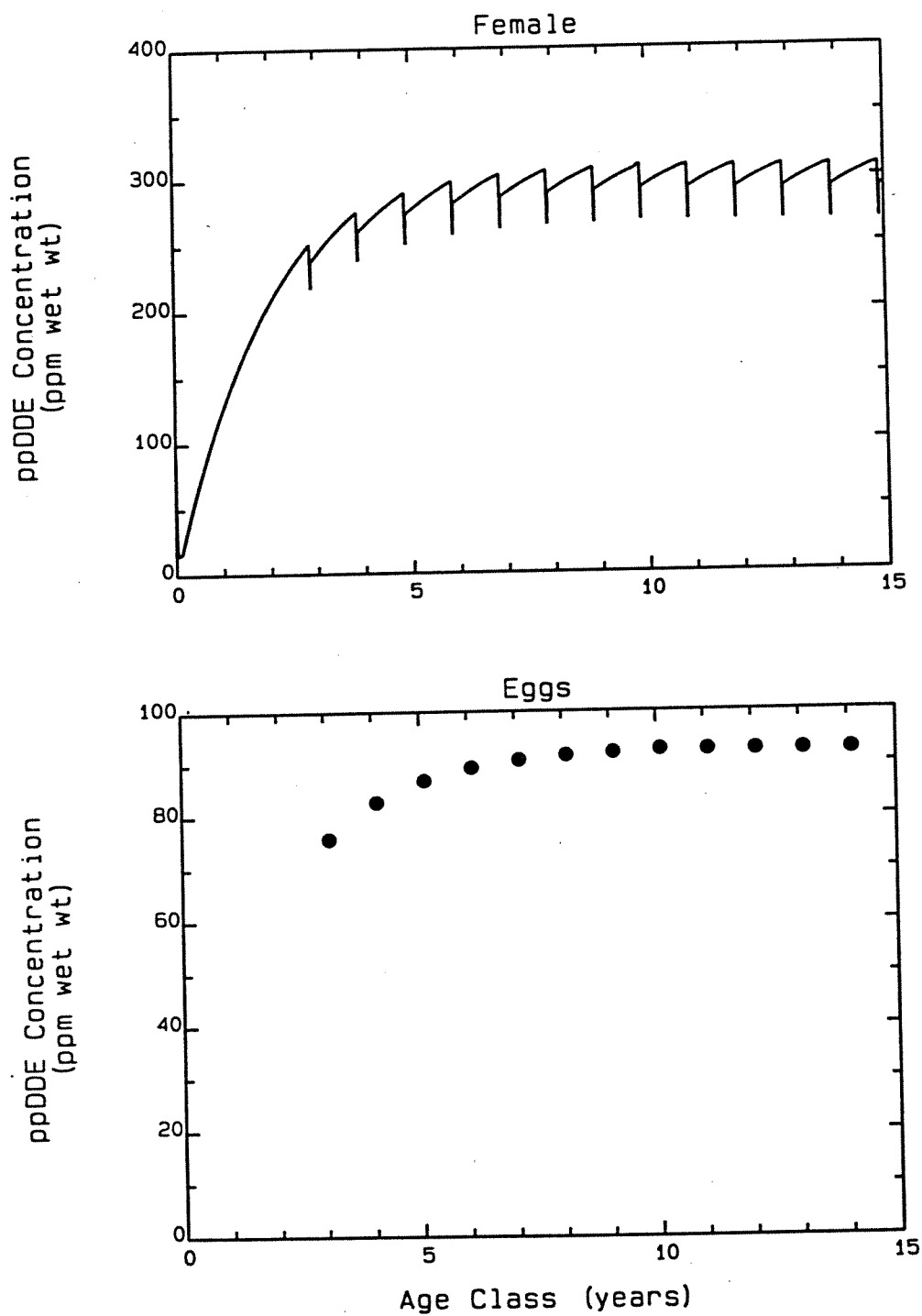
Figure 5-8. Computed p,p'DDE concentrations in female peregrine falcons and their eggs, assuming a constant prey concentration of 4.1 ppm.
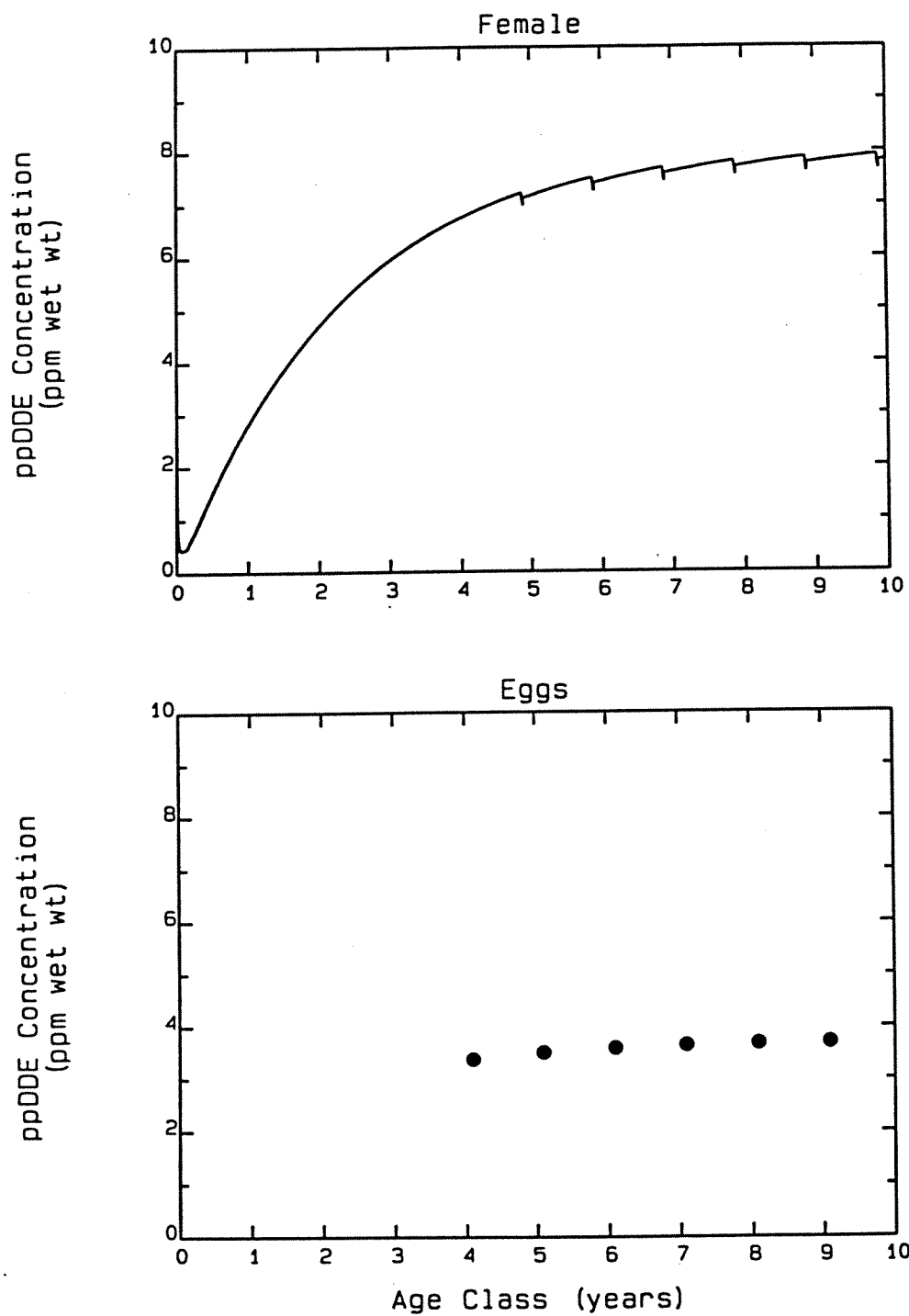
Figure 5-9. Computed p,p'DDE concentrations in female bald eagles and
their eggs, assuming a constant concentration of 0.11 ppm wet weight in the fish prey.

Figure 5-10. Computed p,p'DDE concentrations in female double-crested cormorants and their eggs, assuming a constant level of 0.17 ppm wet weight in the fish prey.
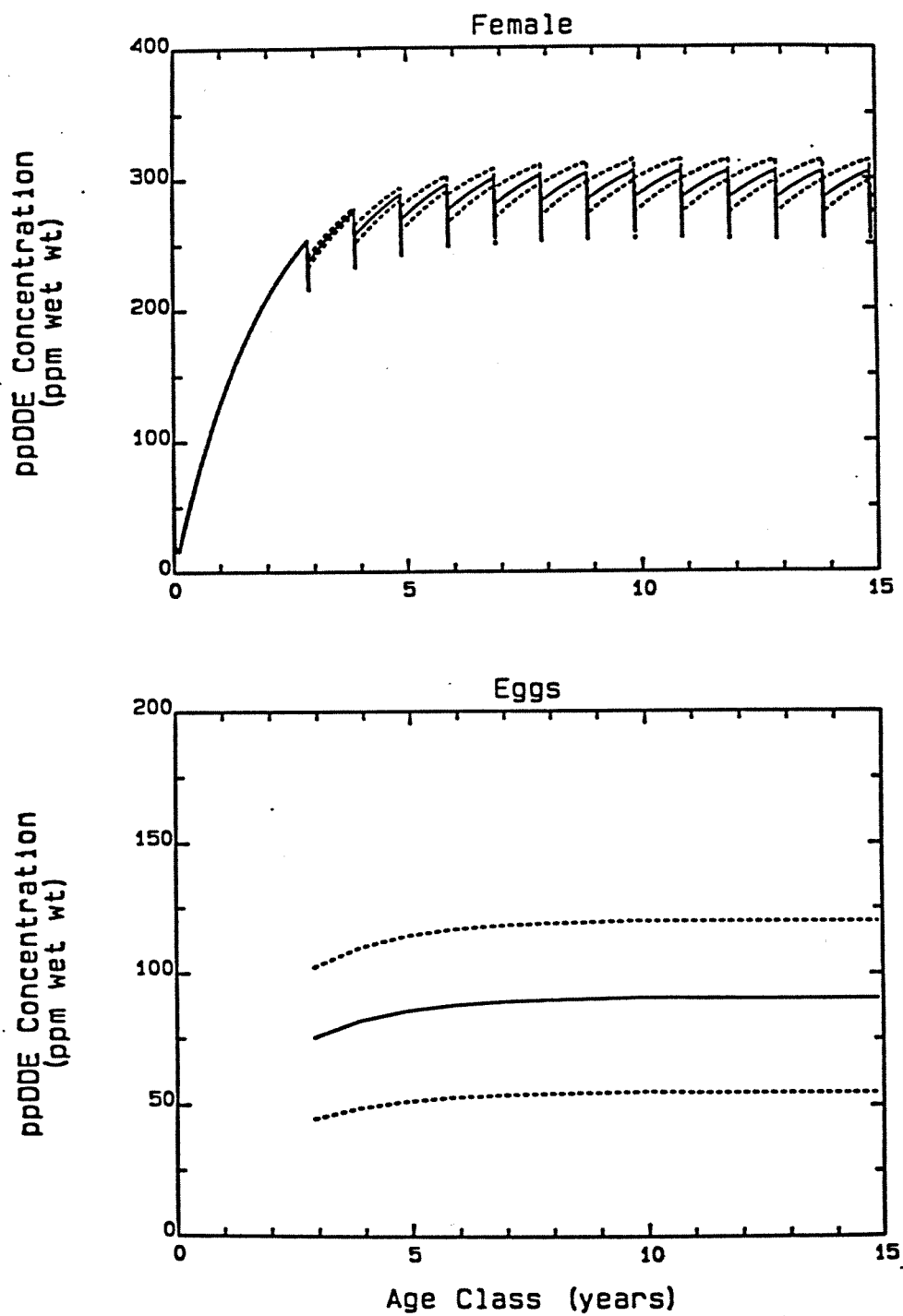
Figure 5-11. The effect of uncertainty in the fraction of dietary lipid
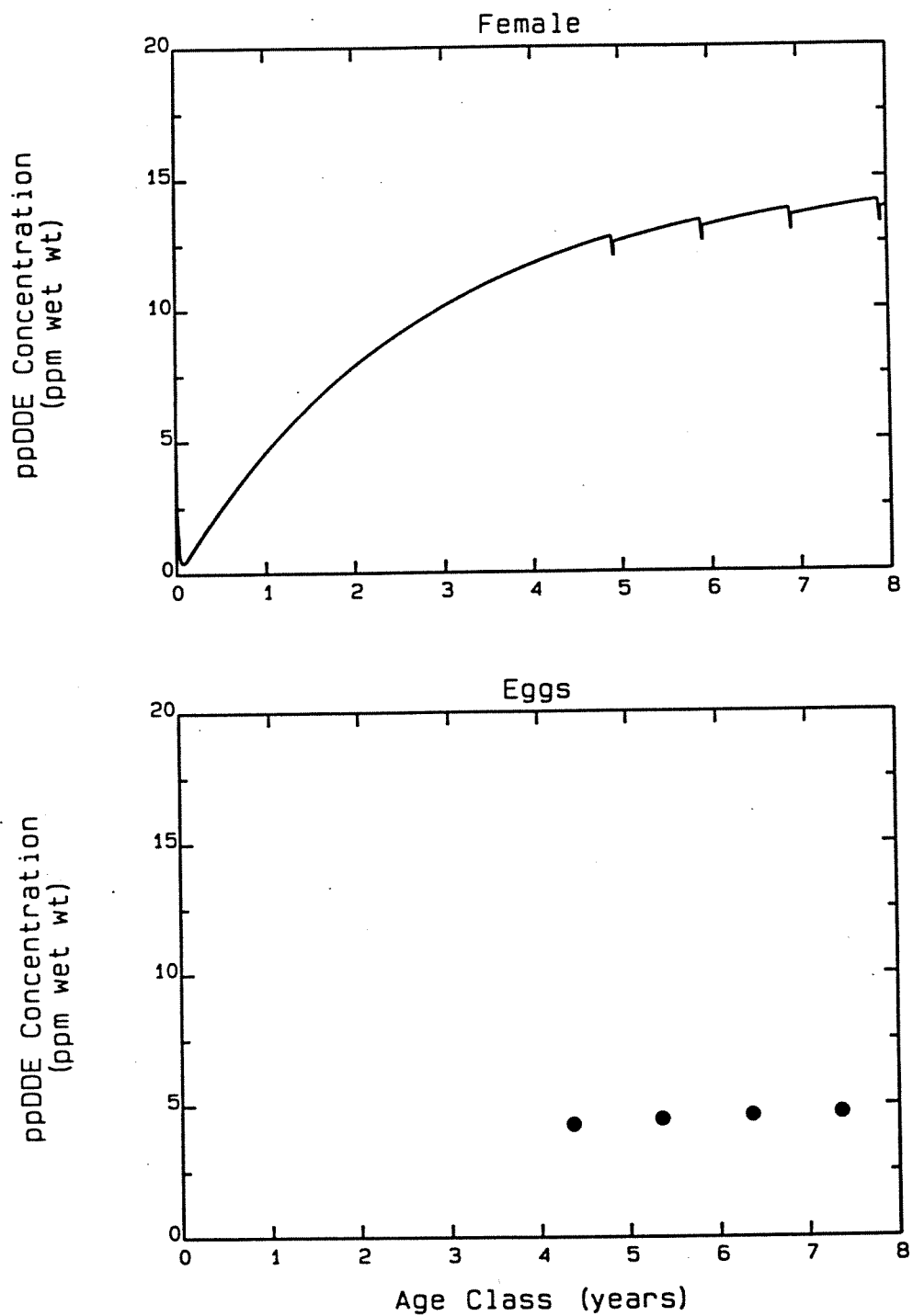in the egg on computed wet weight p,p'DDE concentrations in
female peregrine falcons and their eggs.
Proportion of dietary lipids in the egg = 0.46 (solid line); 0.26 and 0.69 (dashed line).

considered part of the mother until laying. At laying, in the model, the eggs are moved from the mother and the contaminant level in the mother is recomputed. The concentration in the mother post-laying is greater than the concentration in mother + eggs pre-laying, because the eggs have lower wet-weight based contaminant concentrations than the mother (see above). This leads to a small increase in the computed wet-weight body burden at laying, seen for example, in Figures 5-8, 5-9 and 5-10. Following egg-laying, the mother's p,p'DDE level rises throughout the rest of the year (Figures 5-8, 5-9, and 5-10). Concentrations computed in 15-year-old adult male peregrine falcons are on average 12 percent greater than those in females (results not shown). This difference is due to egg production.

Variable Prey Levels. Birds are not exposed to constant doses of contaminants. This variability occurs over a range of time scales, including day-to-day variation in the contaminant levels in prey and season-to-season variation in dietary composition. Two sets of model simulations were performed to test for the impact of daily and seasonal variability in exposure on contaminant levels in the species of interest.

The peregrine falcon model was used to test the effect of day-to-day variation in prey levels. Two simulations were performed. In one, a constant prey concentration was used, and in the other, daily prey concentrations were chosen randomly from a distribution with the same mean as the previous simulation. For the purposes of this test, the distribution of prey levels derived from the Santa Catalina gull p,p'DDE data collected in 1989 was used (Garcelon *et al.* 1989). Note that this distribution was chosen as an example of population variability, not to compare directly with field-measured falcon p,p'DDE levels. A probability plot of the p,p'DDE data from Santa Catalina gulls collected in 1989 is shown on Figure 5-12, top panel. This distribution was used to produce a 10-year record of contaminant levels in food consumed each day (Figure 5-12, bottom panel). This time course of prey contaminant levels was used in the model runs. Computed concentrations in females and eggs were within one percent of the concentrations calculated in the simulation with a constant prey level equal to the average of this simulation. The variation induced in the eggs (coefficient of variation = 0.024) was similar to the simulation with constant prey levels (coefficient of variation = 0.021).

An additional simulation was performed to study the impact on falcons of consuming a diet composed partly of contaminated gulls and partly of clean birds. For this simulation, a diet of 50 percent clean birds and 50 percent gulls from Santa Catalina (1989; Figure 5-1) was used. This resulted in average computed egg concentrations that were 50 percent of the concentration
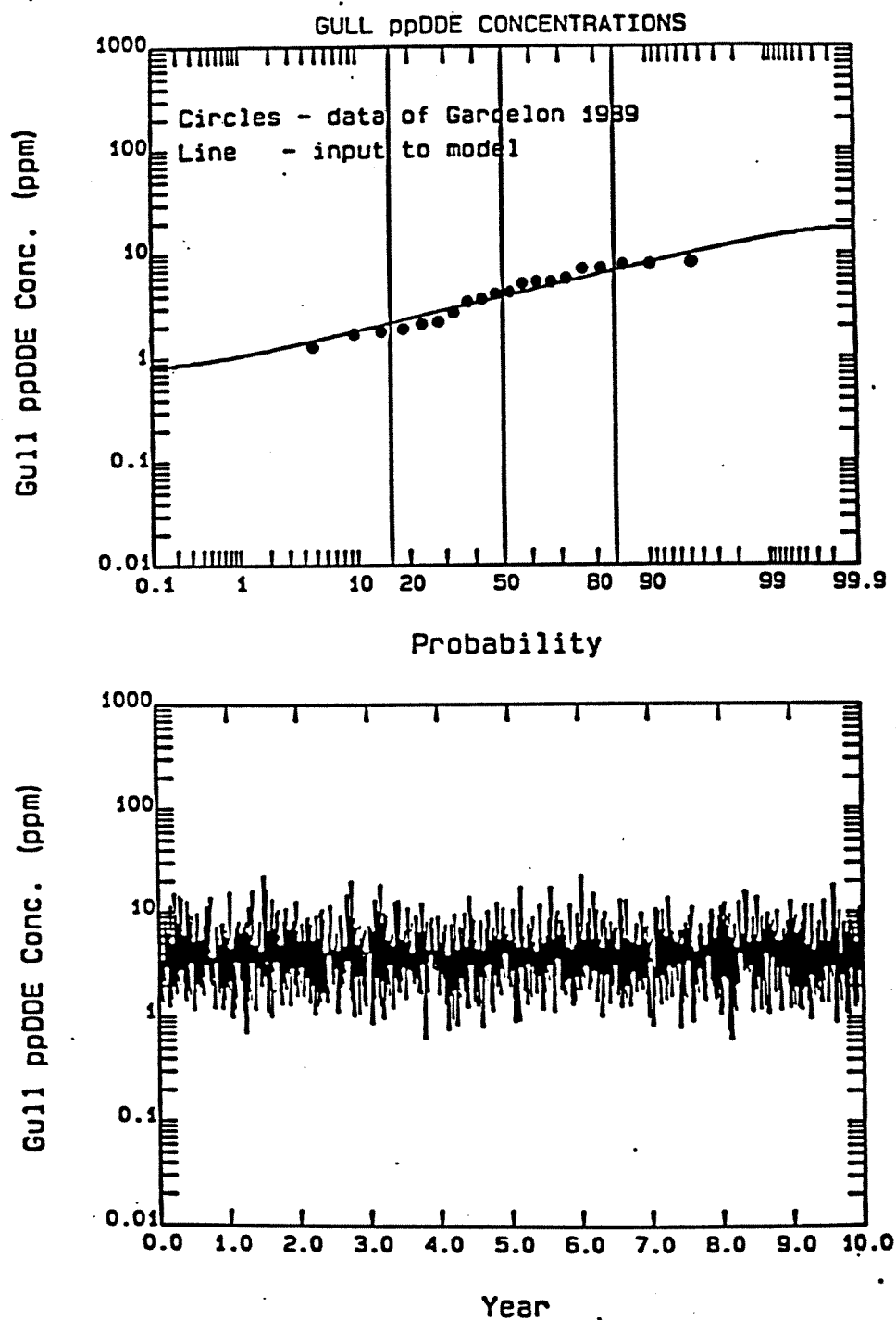
Figure 5-12. Top: Probability distribution of p,p'DDE concentrations in
gulls collected on Santa Catalina Island in 1989;
Bottom: 10 year time course of contaminant levels used in model,
assuming the prey are sampled daily from the distribution in the top panel.

computed in the previous simulation. Thus, the birds integrate exposure levels over time, and average exposure levels can be used in place of the distribution of contaminant levels in prey.

These results do not necessarily extend to the results of occasional consumption of meals of highly contaminated prey during the period of egg yolk production. If the adult bird is exposed to high contaminant levels in prey during that period, the levels in the newly formed eggs may be greater than those predicted by the models as applied here with average prey levels. This is because of the potential for direct transfer of contaminants from the diet to the egg.

The bald eagle model was used to test the effect of seasonal variation in diet. Two simulations were performed. In one, an annual average diet was used (Garcelon *et al.* 1994a and 1994b), and in the other, seasonal dietary information of Garcelon *et al.* (1994a and 1994b) was used. Calculated eagle egg contaminant levels were within 3 percent under the two scenarios. Thus, the annual average dietary composition can be used in place of the seasonally variable information.

5.8   MODEL RESULTS FOR THE SOUTHERN CALIFORNIA BIGHT

Contaminant levels measured in the species of interest are used to establish injury. To establish the pathway for the transfer of p,p'DDE and total PCBs from fish, birds and sea lions of the Southern California Bight to the peregrine falcon, bald eagle and double-crested cormorant, two additional types of field-measured data are required: the composition of the diets of each of the species of interest and the contaminant levels in their prey. Based on these data, the dose received by the birds can be quantified (units of ppm body weight-day).

The final step in establishing pathway is to assess whether it is reasonable to conclude that the species of interest accumulate their contaminant loads from the local species as characterized in the field measurements of dietary composition. To do this, it is necessary to quantify the relationship between the measured contaminant levels in the species of interest (ppm wet egg) and the computed dose (ppm wet-day). It is the purpose of the bioaccumulation models to do this; they provide a mechanistic mathematical framework for computing egg levels in the species of interest from measured diet and prey levels. The computed egg levels can be compared with measured egg levels in the species of interest. A match is evidence supporting the presumed pathway. A mismatch suggests that the dose differs from that estimated using the measurements of diet composition and prey contaminant levels. Such a mismatch would occur

if the species of interest is feeding in locations not previously considered, that is, locations with either higher or lower contaminant levels, or if the measured levels do not accurately reflect the average concentrations in prey within the location sampled.

Peregrine falcon. Computed p,p'DDE concentrations in the females and the eggs of the peregrine falcons from the Southern California Bight are shown in Figure 5-13. Computed p,p'DDE and total PCB concentrations in the eggs of the peregrine falcon are plotted against measured levels in Figure 5-14 (filled and open circles, respectively). The computed and measured egg levels are reported in Table 5-18. The horizontal error bars in Figure 5-14 (for the data) are based on two standard errors of the mean. The vertical error bars in Figure 5-14 (for the computed levels) are based on the range of values of the fraction of dietary lipid in the eggs. The computed egg levels for both p,p'DDE and total PCBs closely match the measured levels, overestimating measured levels by 50 percent at most, and the error bars overlap the 1-to-1 line. Therefore, the computed egg levels are consistent with measured egg levels for both p,p'DDE and total PCBs.

Table 5-18. Concentrations of Contaminants in Bird Eggs - Model Simulations and Data

Species	Model Calculation		Data	
	Best Estimate	Range	Mean	+/-2SE
p,p'DDE				
Peregrine Falcon	30	17 - 41	20	14 - 26
Bald Eagle	37	21 - 51	36	27 - 45
Cormorant Anacapa Is.	10	5.7 - 14	8.0	4.1 - 12
Cormorant S. Barbara Is.	10	5.7 - 14	1.2	0.75 - 1.6
Total PCBs				
Peregrine Falcon	6.3	3.6 - 8.6	5.8	4.4 - 7.3
Bald Eagle	6.8	3.9 - 9.3	8.1	5.7 - 10
Cormorant Anacapa Is.	3.6	2.1 - 4.9	3.1	1.1 - 5.1
Cormorant S. Barbara Is.	3.4	2.0 - 4.7	0.30	0.23 - 0.38

Notes:
No land birds are contaminated in these simulations.
Model range based on range in estimates of the fraction of dietary lipids in eggs.
The cormorants are assumed to feed within 50 km of their breeding island during the breeding season, and throughout the Bight for the rest of the year.
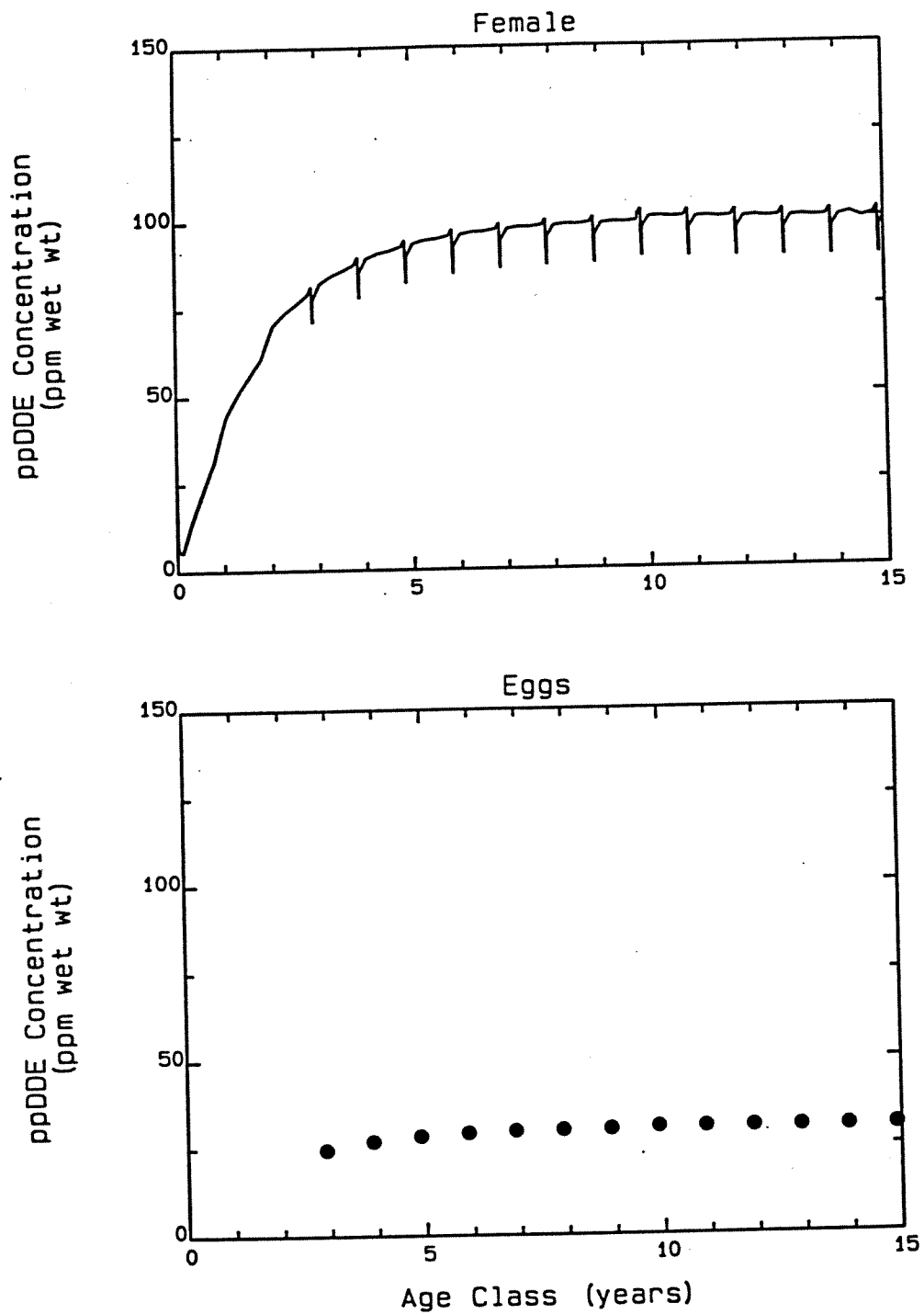
Figure 5-13. Computed p,p'DDE concentrations in female peregrine falcons and their eggs from the Southern California Bight.
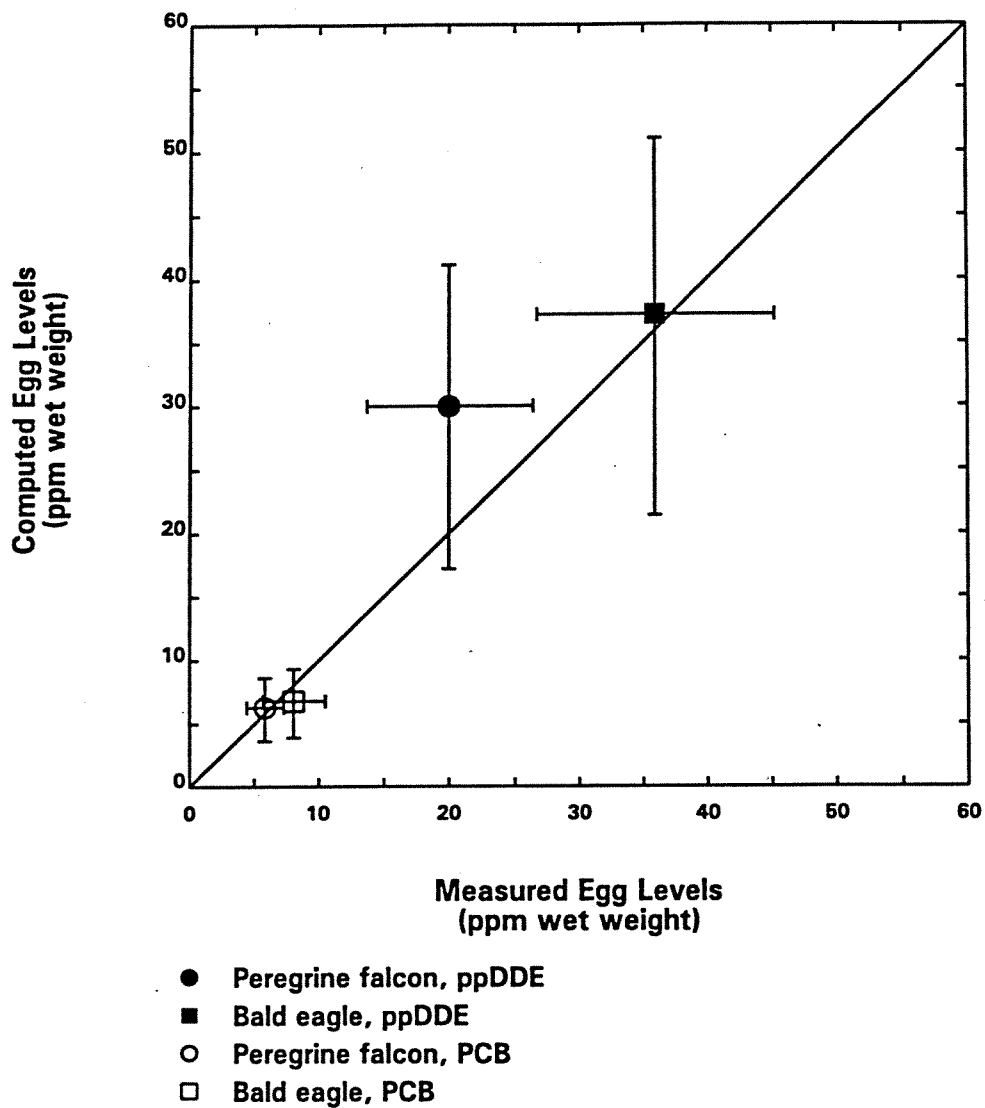
Figure 5-14. Computed and observed contaminant concentrations in the eggs of the peregrine falcon and bald eagle from the Southern California Bight (ppm wet weight). Horizontal error bars represent +/- two standard errors of the mean. Vertical error bars represent the range of the fraction of dietary lipid in the egg.

·One uncertainty in estimating contaminant dose was the contaminant level in migratory land birds. The simulations discussed above were performed assuming that migratory land birds contained no contaminants. An additional model simulation was performed with contaminated migratory land birds. The computed egg levels were within 10 percent of the first simulation for both contaminants. Thus, whether or not the land birds are contaminated makes little difference to the results.

Bald eagle. Computed p,p'DDE concentrations in the females and the eggs of the bald eagle from the Southern California Bight are shown in Figure 5-15. Computed p,p'DDE and total PCB concentrations in the eggs of the bald eagle are plotted against measured levels in Figure 5-14 (filled and open squares, respectively). The computed and measured egg levels are reported in Table 5-18. The error bars in Figure 5-14 have the same meaning as for the peregrine falcon. The computed egg levels are within 5 percent (p,p'DDE) and 20 percent (PCBs) of the measured levels, and the error bars overlap the 1-to-1 line. Therefore, the computed egg levels are consistent with measured egg levels for both p,p'DDE and total PCBs.

As for the peregrine falcon, the effect of uncertainty in the degree of contamination in migratory land birds was explored in an additional simulation. The computed egg levels were within 2 percent of the first simulation for both p,p'DDE and total PCBs. Thus, whether or not the land birds are contaminated makes little difference to the results.

Another issue is the degree to which contaminant levels might differ between eagles hatched in the Southern California Bight and introduced eagles. Between 1980 and 1986 young eagles were obtained from nests in northern California, Washington and British Columbia and reintroduced onto Santa Catalina Island (Risebrough 1987). A simulation was performed to study the accumulation of contaminants in a juvenile bald eagle introduced onto Santa Catalina Island. For the first year of life in the model, the eagle was assumed to consume uncontaminated food (Risebrough 1987).

The time course of p,p'DDE concentration in a single introduced bald eagle is shown in Figure 5-16. The contaminant level begins to rise after the first year, when the young eagles begin to catch fish and birds and eat sea lion blubber. By the fifth year, when the birds begin

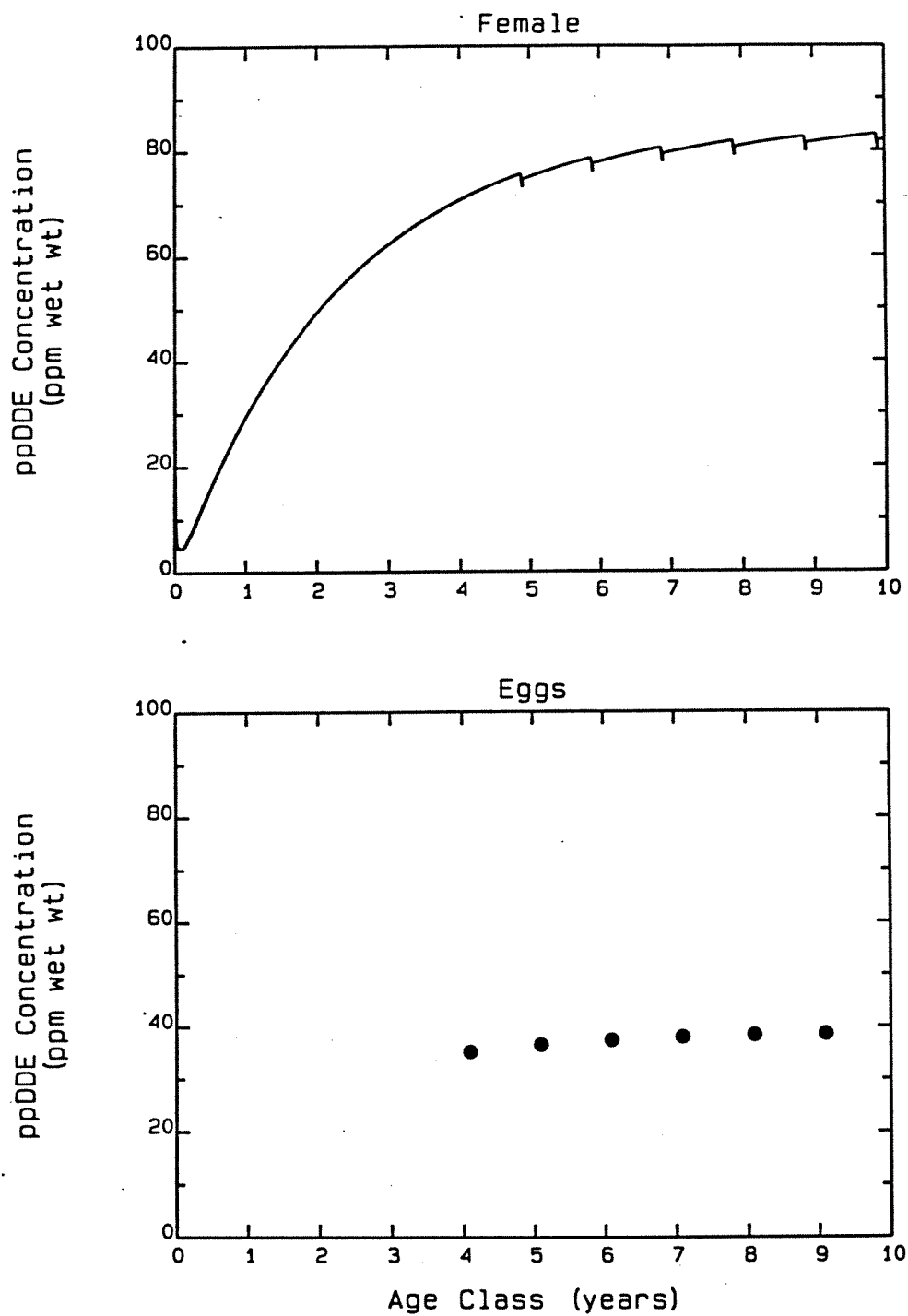Figure 5-15. Computed p,p'DDE concentrations in female bald eagles and their eggs from Santa Catalina Island.
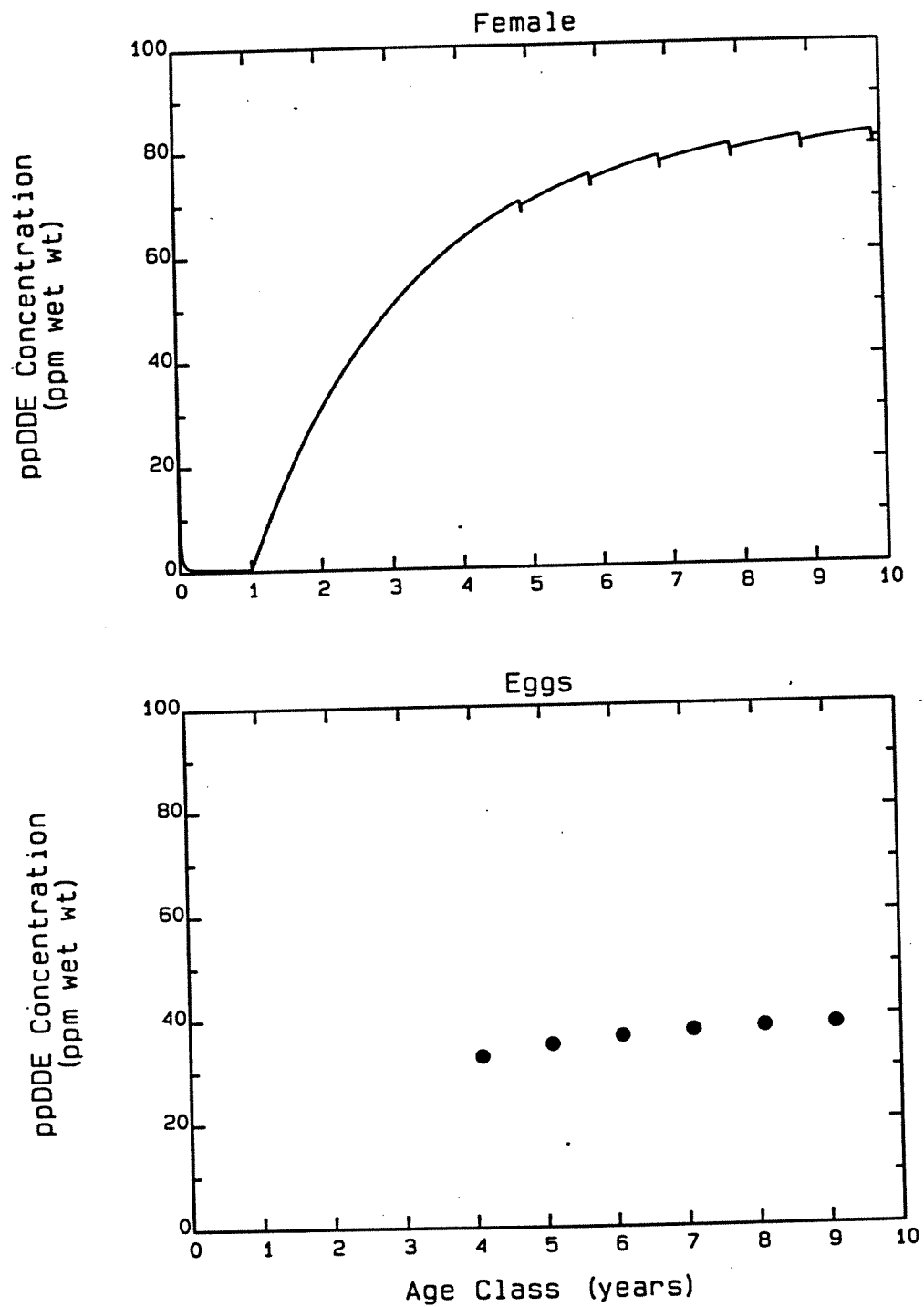
Figure 5-16. Computed p,p'DDE concentrations in female bald eagles and their eggs introduced to Santa Catalina Island.

to breed, the contaminant level is within 25 percent of the simulation for birds hatched in the Southern California Bight. The effect on eggs produced by older birds is less than this. Thus, the effect of consuming clean food during the first year of life on the concentrations achieved in the eggs throughout the bird's reproductive period is considerably less than 25 percent.

Double-crested cormorant. The estimated concentration in Anacapa cormorants is shown in Figure 5-17. The concentrations increase during the first four years of life, because the birds are feeding throughout the Bight, which includes the areas with greater contaminant levels. Thereafter, the annual cycle includes the effects of egg laying and a change in diet between the breeding and non-breeding seasons (within 50 km of the island during the breeding season and throughout the Bight during the rest of the year).

Model results for the double-crested cormorants on Anacapa Island are within 25 percent (p,p'DDE) and 20 percent (PCBs) of the average of the data; the model overestimates the average of the data (Table 5-18, Figure 5-18). For Santa Barbara Island, the model overestimates the data by approximately ten-fold (Table 5-18, Figure 5-18). Additional model simulations were performed to explore the effect of assuming that the cormorants feed exclusively near the breeding island. This scenario resulted in model computations for Anacapa Island that underestimated the data by 40 and 30 percent for p,p'DDE and PCBs, respectively. The model results for Santa Barbara Island were still 3 to 4 times greater than the data average (Table 5-19, Figure 5-18). These results suggest that the double-crested cormorants on Anacapa Island are feeding to some degree in the more highly contaminated regions of the Southern California Bight, but that the cormorants on Santa Barbara Island are feeding more extensively outside of the more contaminated regions of the Southern California Bight. It is also possible that our estimates of fish concentrations near Santa Barbara Island are greater than the true concentrations.

Conclusion. Overall, the computed and measured egg contaminant concentrations are quite similar. Considering both chemicals in the peregrine falcon, the bald eagle and the double-crested cormorant on Anacapa Island, in five of six comparisons, computed and measured egg concentrations differ by at most 25 percent. In the other comparison, p,p'DDE in the peregrine falcon, the difference is 50 percent. This relatively larger difference may be due to imprecision in measurements of p,p'DDE levels in the peregrine falcon or in its prey. The overall conclusion is that the dietary composition developed from field measurements during the Southern California Bight Damage Assessment characterizes the contaminant sources to the
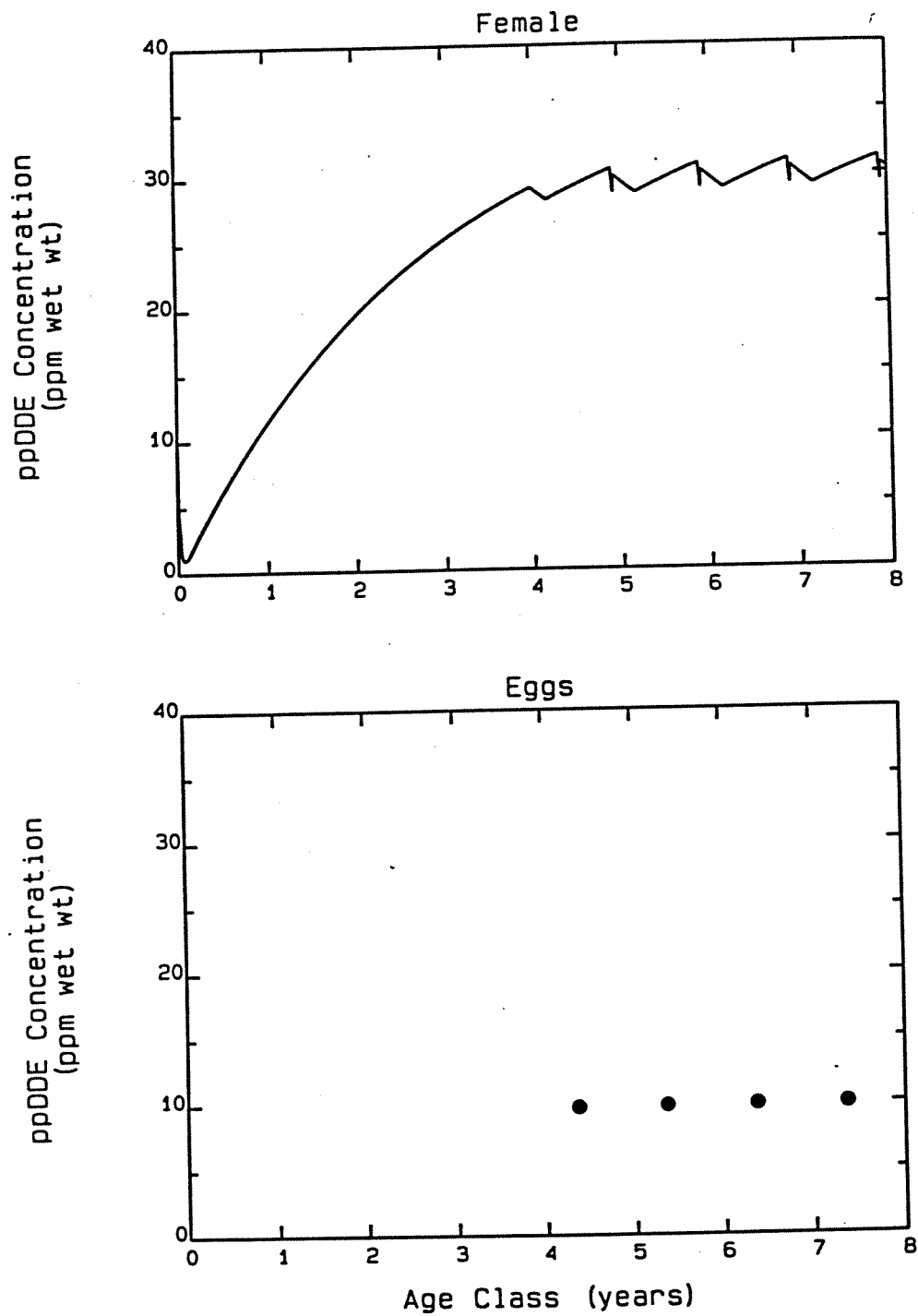
Figure 5-17. Computed p,p'DDE concentrations in female double-crested cormorants nesting on Anacapa Island and their eggs. The birds feed within 50 km of the island during the breeding season and throughout the Bight during the rest of the year.
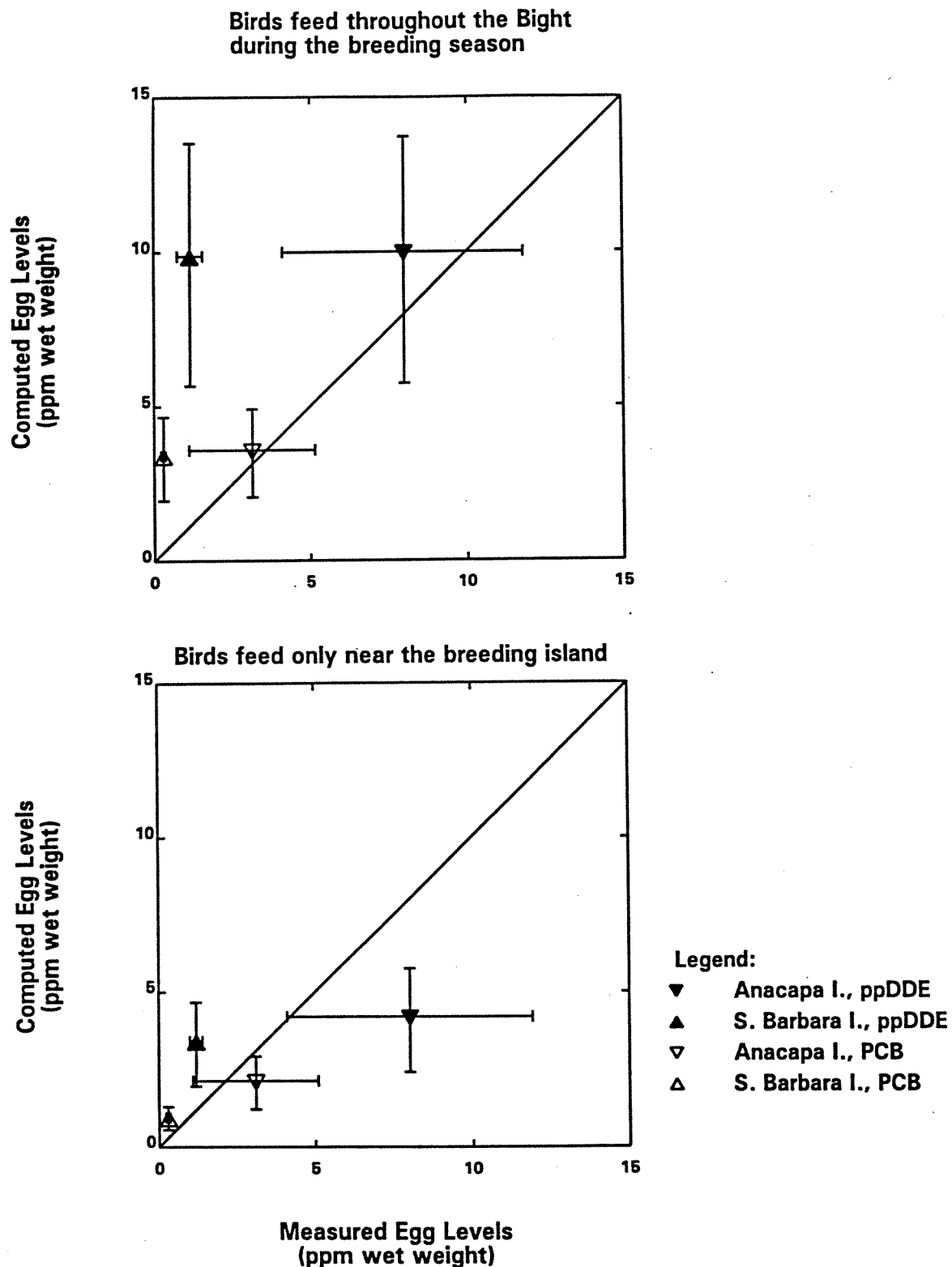
Figure 5-18. Computed and observed contaminant concentrations in the eggs of the double-crested cormorant from the Southern California Bight (ppm wet weight). Horizontal error bars represent +/- two standard errors of the mean. Vertical error bars represent the range of the fraction of dietary lipid in the egg. Values in top panel from Table 5-18; values in bottom panel

double-crested cormorant on Anacapa Island, the peregrine falcon, and the bald eagle realistically.

In contrast, the computed egg concentrations for both chemicals in the double-crested cormorant on Santa Barbara Island are 8 to 11 times greater than measured levels. This is likely due to a bias in the assumed feeding behavior of the cormorants, but may also be affected by unrepresentative sampling of these birds.

Table 5-19. Concentrations of Contaminants in Double-Crested Cormorant Eggs - Model Simulations for Birds Feeding near the Breeding Island and Data

Island	Model Calculation		Data	
	Best Estimate	Range	Mean	+/-2SE
p,p'DDE				
Anacapa Is.	4.2	2.4 - 5.8	8.0	4.1-12
S. Barbara Is.	3.4	2.0 - 4.7	1.2	1.0-1.4
total PCBs				
Anacapa Is.	2.1	1.2 - 2.9	3.1	1.1-5.1
S. Barbara Is.	1.0	0.55 - 1.3	0.30	0.23-0.37

Note: Model range based on range in estimates of the relationship between lipid-based concentration in eggs and whole body.

5.9 DOSE CALCULATIONS

The proportion of the total dose of p,p'DDE and PCBs received by the bald eagle and the peregrine falcon that is attributable to sources within the Southern California Bight was estimated. The proportion of each prey type in the diet on an energy basis was multiplied by the concentration of contaminants in that prey, giving an estimate of the proportional contribution of each prey type to the total dose. These values were then multiplied by estimates of the proportion of the contaminants in each prey type considered to have originated in the Southern California Bight. The estimates of the proportion of the dose originating in the Southern California Bight may be conservative because of the following:

●   The value of 50 percent for the proportion of the contaminant loads in California gulls, Heermann's gulls, Bonaparte's gulls and other water birds that originates outside the Southern California Bight is based on the following assumptions: (1) the birds spend approximately 50 percent of their time outside the Southern California Bight, and (2) the concentration of contaminants in their prey outside

the Southern California Bight is approximately equal to that found in the Southern California Bight.

- All contamination in migratory land birds originates outside the Southern California Bight in the dose calculation.

The percentage of the total contaminant dose to the peregrine falcon that originates in the Southern California Bight is estimated to be:

- 77 and 74 percent for p,p'DDE and total PCBs (both resident and migratory land birds uncontaminated; Table 5-20).

- 75 and 69 percent for p,p'DDE and total PCBs (residents uncontaminated; migratory land birds contaminated; Table 5-21).

If the contaminant levels in prey of the migratory gulls outside of the Bight is less than within the Bight, then the overall proportion of the dose that is received within the Bight is greater. For example, if the concentration in prey of these gulls outside the Bight equals one-half the concentration within the Bight, then the peregrine falcon receives 82% and 77% of it total dose of p,p'DDE and PCBs from within the Bight.

The percentage of the total contaminant dose to the bald eagle that originates in the Southern California Bight is estimated to be between 91 and 93 percent for both p,p'DDE and total PCBs, whether land birds are considered to be clean or contaminated (Tables 5-22 and 5-23).

Table 5-20. Dose Calculations for Peregrine Falcon - No Land Birds Contaminated

Species	Percent of load from within SCB	p,p'DDE in the peregrine diet		total PCBs in the peregrine diet	
		Percent of dose (energy basis)	Percent of diet from within SCB	Percent of dose (energy basis)	Percent of diet from within SCB
Western gull	100	29	29	32	32
California gull	50	17	8	18	9
Heermann's gull	50	5	2	5	3
Bonaparte's gull	50	4	2	5	2
Cassin's auklets	100	25	25	16	16
Other water birds	50	21	10	24	12
Land birds - resident	0	0	0	0	0
Land birds - migratory	0	0	0	0	0
Sum			77		74

Table 5-21. Dose Calculations for Peregrine Falcon - Migratory Land Birds Contaminated

Species	Percent of load from within SCB	p,p'DDE in the peregrine diet		total PCBs in the peregrine diet	
		Percent of dose (energy basis)	Percent of diet from within SCB	Percent of dose (energy basis)	Percent of diet from within SCB
Western gull	100	28	28	29	29
California gull	50	16	8	17	9
Heermann's gull	50	5	2	5	3
Bonaparte's gull	50	4	2	4	2
Cassin's auklets	100	24	24	15	15
Other water birds	50	20	10	22	11
Land birds - resident	0	0	0	0	0
Land birds - migratory	0	3	0	7	0
Sum			75		69

Table 5-22. Dose Calculations for Bald Eagle - No Land Birds Contaminated

Species	Percent of load from within SCB	p,p'DDE in the eagle diet		total PCBs in the eagle diet	
		Percent of dose (energy basis)	Percent of dose from within the SCB	Percent of dose (energy basis)	Percent of dose from within the SCB
Fish and invertebrates	100	8	8	20	20
Sea lions	100	59	59	45	45
Other mammals	100	0	0	0	0
Western gull	100	19	19	19	19
Other gulls	50	4	2	4	2
Water birds	50	9	5	12	6
Land birds-resident	0	0	0	0	0
Land birds-migratory	0	0	0	0	0
Sum			93		92

Table 5-23. Dose Calculations for Bald Eagle - Migratory Land Birds Contaminated

Species	Percent of load from within SCB	p,p'DDE in the eagle diet		total PCBs in the eagle diet	
		Percent of dose (energy basis)	Percent of dose from within the SCB	Percent of dose (energy basis)	Percent of dose from within the SCB
Fish and invertebrates	100	8	8	20	20
Sea lions	100	59	59	45	45
Other mammals	100	0	0	0	0
Western gull	100	18	18	19	19
Other gulls	50	4	2	4	2
Water birds	50	9	5	12	6
Land birds-resident	0	0	0	0	0
Land birds-migratory	0	<1	0	<1	<1
Sum			92		92

SECTION 6

CONTAMINANT LEVELS IN FISH POPULATIONS OF THE SOUTHERN CALIFORNIA BIGHT: PROPORTIONS EXCEEDING SPECIFIED VALUES

6.1   OBJECTIVE

The work presented in Sections 2 and 3 established that the contaminants in fish from the Palos Verdes Shelf area originate in local sediments and water. This was accomplished by comparing average measured concentrations with concentrations computed by the bioaccumulation model. Here, the objective is to explore the potential for injury in white croaker, Dover sole and kelp bass in each of several regions of the Southern California Bight. The strategy is to estimate the proportion of each species in each model segment that contain total DDT or total PCB levels that are greater than specified levels. The specified levels include the FDA limits for PCBs and total DDTs, a critical level in fish ovaries that is related to fish reproduction (Hose and Cross 1994), and critical levels in fish determined based on an analysis of cancer risk by Pollock *et al.* (1991) (Table 6-1). This work was based solely on analyses of data and did not rely on and of the model results.

Table 6-1. Key Contaminant Concentrations in Fish

Chemical	Tissue	Specified Value (ppm wet weight)
Total DDT	Ovary	4
Total DDT	Muscle	5
Total PCB	Muscle	2
Total DDT	Muscle	0.1
Total PCB	Muscle	0.1

This analysis involved developing a database of contaminant concentrations in fish from the Southern California Bight and a computer program to analyze the data. The specified values are tissue-specific, and data are not always available for the tissue of interest. Therefore, the program includes conversion factors whereby values measured in one tissue can be converted to equivalent concentrations in another tissue. The values of the conversion factors were computed based on analyses of studies conducted using fish from the Southern California Bight as well as other locations.

For the specified muscle concentrations, only muscle data were used. Data on ovary contaminant levels are scarce compared to the data on muscle and liver contaminant levels. The only fish ovary data in the HydroQual database are 131 values for white croaker; in contrast, the database includes more than 1,000 records for contaminants in muscle and liver of many species of fish. Therefore, for the specified ovary level, measurements conducted using muscle, liver and whole-body samples were converted to equivalent ovary concentrations. Species-specific conversion factors were used whenever possible. When not possible, generic values were developed based on published information.

Muscle, liver and whole-body data collected in the Southern California Bight between 1985 and 1995 were used in this analysis, because previous analyses have shown that over that time there was little or no trend in contaminant levels (Chapter 2).

6.2   MUSCLE TISSUE RESULTS

Calculations were performed for total DDT and total PCB. Where only p,p'DDE data were available, these were divided by 0.87 to give equivalent total DDT values, because p,p'DDE represents an average of 87% of total DDT in all fish in the HydroQual database. All calculations of muscle contaminant levels were performed using concentrations measured in muscle tissue on a wet weight-basis.

To illustrate the method of analysis, a probability plot for total DDT in the muscle tissue of white croaker caught in segment 7 is shown in Figure 6-1. The cumulative probability plot is a statistical representation of the data. The x-axis shows the probability that a data point within a data distribution will be at or below the values plotted on the y-axis. Specifically for Figure 6-1, the x-axis represents the probability that the total DDT concentration in the muscle tissue of a fish caught within segment 7 will be at or below the values on the y-axis.

The proportions of each species in each location with calculated tissue contaminant concentrations greater than the specified value were calculated as follows. First, a regression line was fit through each probability plot; this is illustrated in Figure 6-1. Such a regression line on a log-probability plot establishes the parameters of the log normal distribution (mean and standard deviation). Next, the regression line was used to compute the proportion of each sample distribution with concentrations greater than the specified value. This illustrated by the intersection of the solid regression line in Figure 6-1 with the horizontal dashed line representing
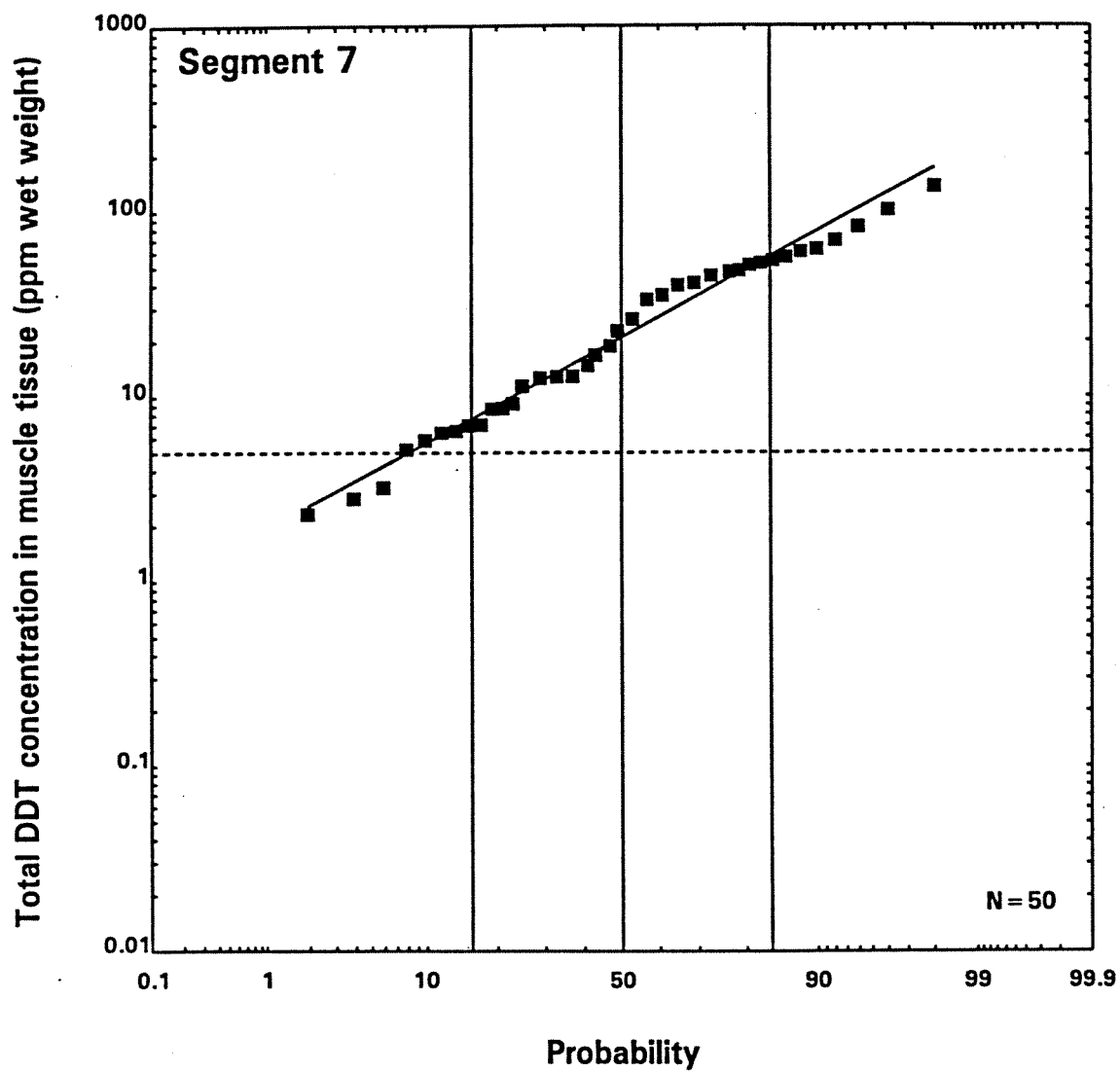
Figure 6-1.   Total DDT concentration (ppm wet weight) in muscle tissue of white croaker from Segment 7, 1985 to 1995.

the specified value, in this case 5 ppm; at the point of intersection, the proportion of the population with concentrations lower than the specified value is read off the x-axis. Subtracting this number from one hundred results in the proportion of the fish population within this segment which exceeds 5 ppm. In this example, 92 percent of the white croaker in segment 7 contain total DDT levels greater than 5 ppm wet weight in muscle.

The above analysis was performed on muscle tissue for the values presented in Table 6-1 for each HydroQual segment for which there was two or more observations. Probability plots of total DDT and total PCB for white croaker, kelp bass, and Dover sole in each HydroQual segment are presented in Appendix E.

Figures 6-2 through 6-5 present the results for total DDT and total PCB at each of the above key values. These bar charts show the proportion of the fish population within each segment which exceed a key value. The segments are ordered on the y-axis according to the distance from the Whites Point Outfall. The number of samples is indicated to the right of each bar. In general, the segments with the highest proportion of exceedances are those closest to the outfall. For example, any segment in which total DDT in white croaker muscle tissue exceeds 5 ppm for more than 40% of the population is within six kilometers of the outfall (Figure 6-2). It should be noted that there was only one observation for kelp bass in segment 8, therefore no regression analysis could be performed.

6.3   OVARY CONCENTRATIONS

The analysis for the key ovary concentration differed from the key muscle concentration because muscle and liver data had to be converted to equivalent ovary levels.

6.3.1 Tissue Conversion Factors

Method

Hydrophobic compounds such as DDTs and PCBs accumulate in the lipid fraction of tissues. This causes the tissue levels expressed on a lipid basis to be more similar than levels expressed on a wet weight-basis. For example, the average wet weight-based liver/muscle ratio of p,p'DDE concentrations in black surfperch (Embiotoca jacksoni) from the Southern California Bight is 32, and the average lipid-based ratio is 1.2.

Distance from LA Outfall
(kilometers)

North
160
160
108
72
44
44
32
26
14
11
6
4
2
0
2
4
11
22
35
37
44
92
South

White croaker

Kelp bass

Dover sole

Percentage which exceeds 5 ppm wet weight for total DDT

HydroQual segment number
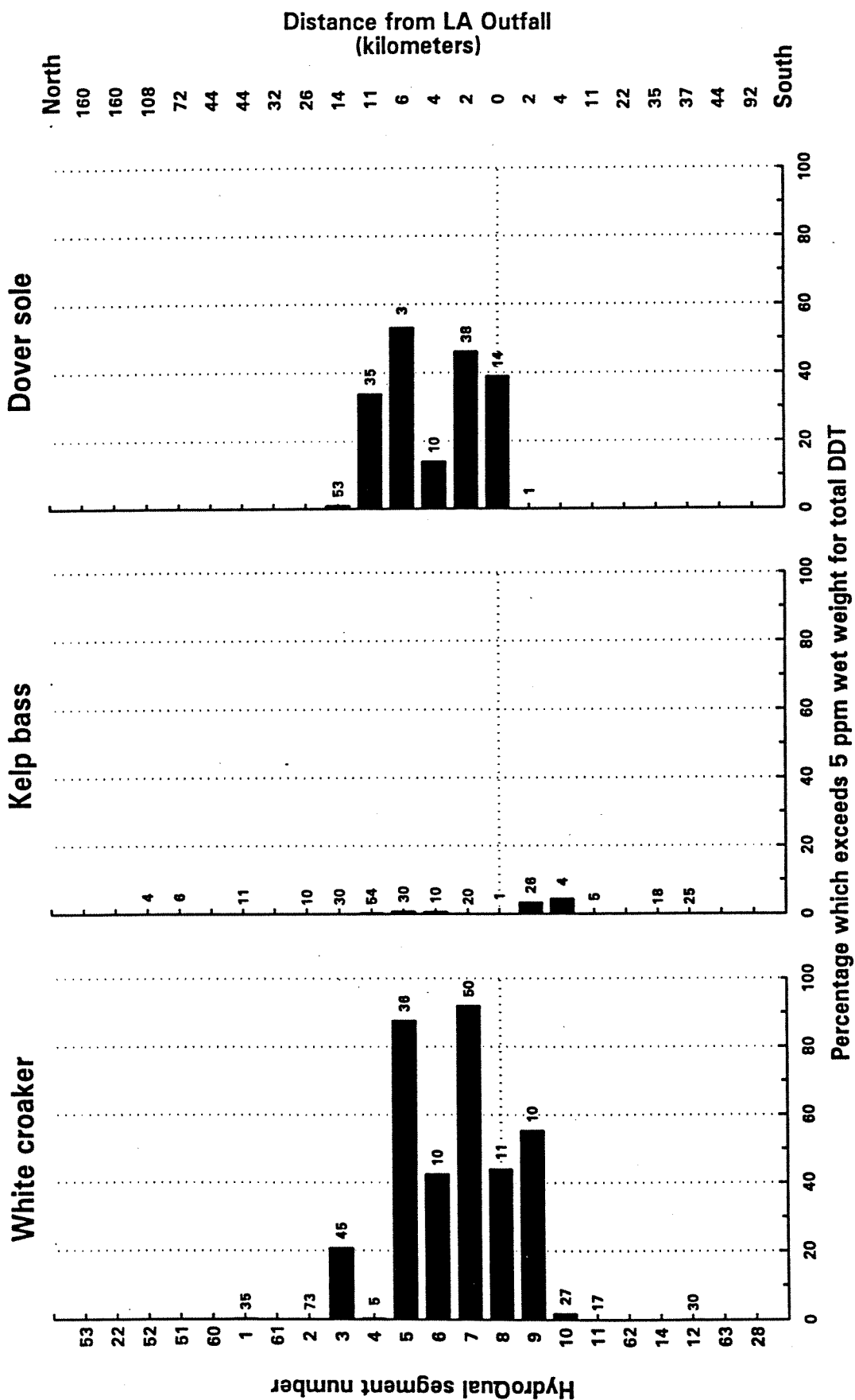
PVS: Segments 3 - 10.
West Outfall: Segment 8.

Figure 6-2.   Computed percentage of fish with muscle total DDT concentration greater than 5 ppm wet weight. The number of samples per segment is posted to the right of each bar.

Figure 6-3. Computed percentage of fish with muscle total PCB concentration greater than 2 ppm wet weight. The number of samples per segment is posted to the right of each bar.

Figure 6-4. Computed percentage of fish with muscle total DDT concentration greater than 0.1 ppm wet weight. The number of samples per segment is posted to the right of each bar.
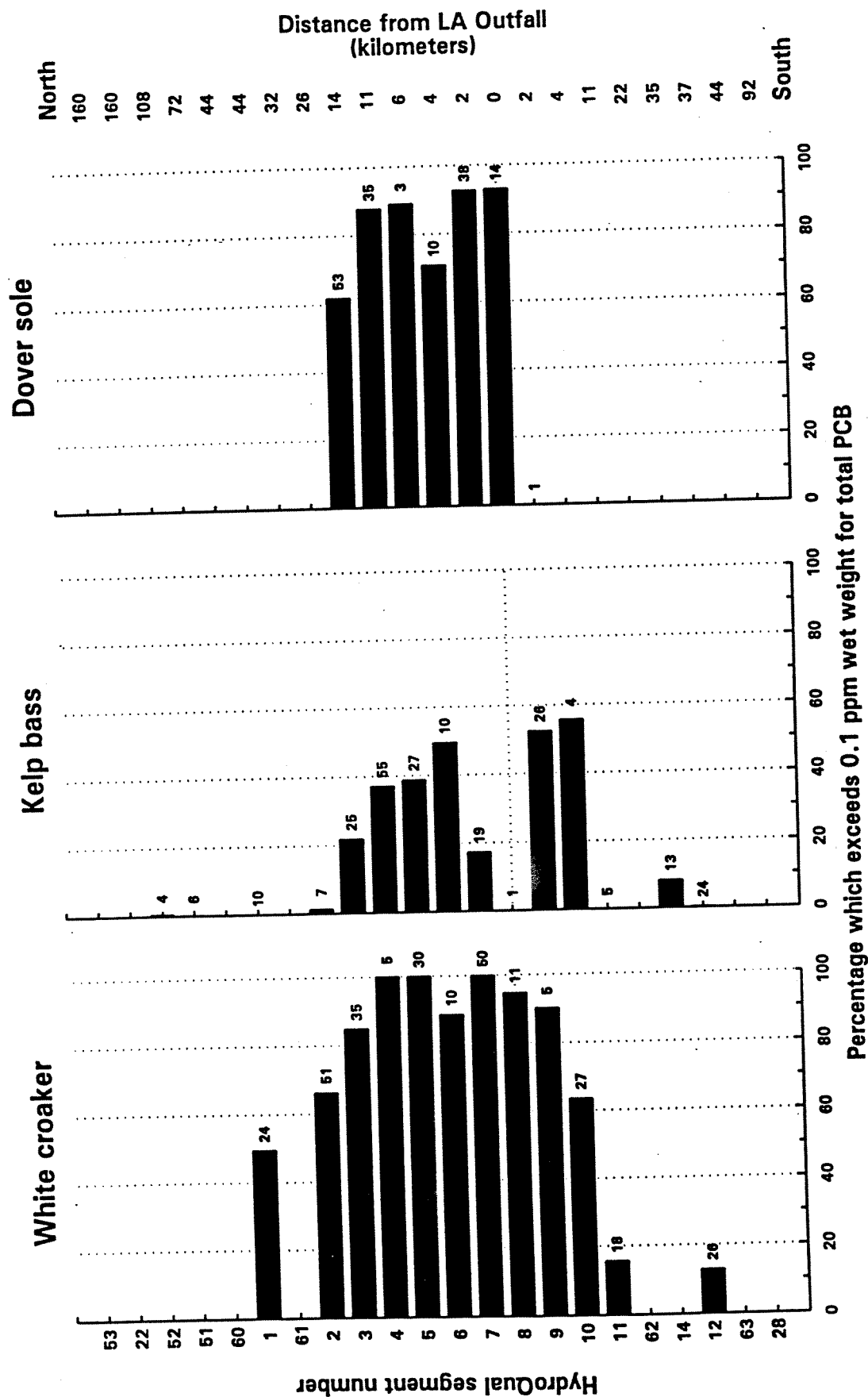
Figure 6-5.   Computed percentage of fish with muscle total PCB concentration greater than 0.1 ppm wet weight. The number of samples per segment is posted to the right of each bar.

Therefore, to estimate relationships between contaminant levels in different tissues, lipid-based concentrations were used. Muscle and whole body lipid-based contaminant levels were assumed to be equal. To convert liver data to equivalent muscle contaminant levels, the relationship between muscle and liver tissue contaminant levels was examined. Then the relationship between muscle and ovary levels was examined using published information for several species.

Muscle/Liver Relationship

To develop a relationship between liver and muscle lipid-based contaminant levels, total PCB and total DDT concentrations measured in these two tissues were compiled for several species using published literature, including several measurements from the Southern California Bight. Only studies in which liver and muscle were sampled from the same fish were used. The lipid-based concentration ratios calculated from the results of each study are given in Table 6-2. The units of these values are:

$$\frac{\mu g \text{ contaminant/g lipid in liver}}{\mu g \text{ contaminant/g lipid in muscle}} \qquad (6\text{-}1)$$

Also tabulated in Table 6-2 are the ratios of lipid contents in the two tissues. The units of these values are:

$$\frac{g \text{ lipid /g wet weight in liver}}{g \text{ lipid/g wet weight in muscle}} \qquad (6\text{-}2)$$

Measured lipid-based liver/muscle contaminant ratios vary over almost two orders of magnitude (0.69 to 40.9; Table 6-2). To explore the causes for this variation, the contaminant ratios were plotted against the ratio of lipid contents (Figure 6-6). The two values in the upper left of Figure 6-6 (shortspine thornyhead; contaminant ratios = 40.9 for p,p'DDE and 9.7 for total PCB) were based on a small sample size (n=5) and had coefficients of variation larger than any other data (4.2 and 3.9), as well as the smallest lipid content ratio (0.093). Therefore, these two points were not used in the analysis. Without these two points, the liver/muscle contaminant

Table 6-2. Liver/Muscle Contaminant Ratios for Fish

Species	Location	Chemical	Ratios		Reference
			Lipid Content	Lipid-based Contaminant Concentration	
Baltic herring	German Baltic coast	DDE	0.308	0.816	Hansen et al. 85
Baltic herring	German Baltic coast	pcb 1260	0.308	0.709	Hansen et al. 85
dab	coastal France	pcb 138,153	2.990	0.694	Loizeau&Abarnou 94
cod	Western Baltic	pcb	104.167	2.400	Schneider 82(5)
black surfperch	SCB	total PCBs	31.530	1.050	Risebrough 87
black surfperch	SCB	p,p'DDE	31.530	1.220	Risebrough 87
kelp bass	SCB	total PCBs	12.320	1.470	Risebrough 87
kelp bass	SCB	p,p'DDE	12.320	1.550	Risebrough 87
white croaker	SCB	total PCBs	6.800	1.590	Risebrough 87
longspine thornyhead	SCB	total PCBs	11.410	1.620	Risebrough 87
white croaker	SCB	p,p'DDE	6.800	1.800	Risebrough 87
longspine thornyhead	SCB	p,p'DDE	11.410	1.870	Risebrough 87
spotted ratfish	SCB	total PCBs	60.020	6.020	Risebrough 87
shortspine thornyhead	SCB	total PCBs	0.093	9.710	Risebrough 87
spotted ratfish	SCB	p,p'DDE	60.020	13.700	Risebrough 87
shortspine thornyhead	SCB	p,p'DDE	0.093	40.900	Risebrough 87

Note: Units are given in the text
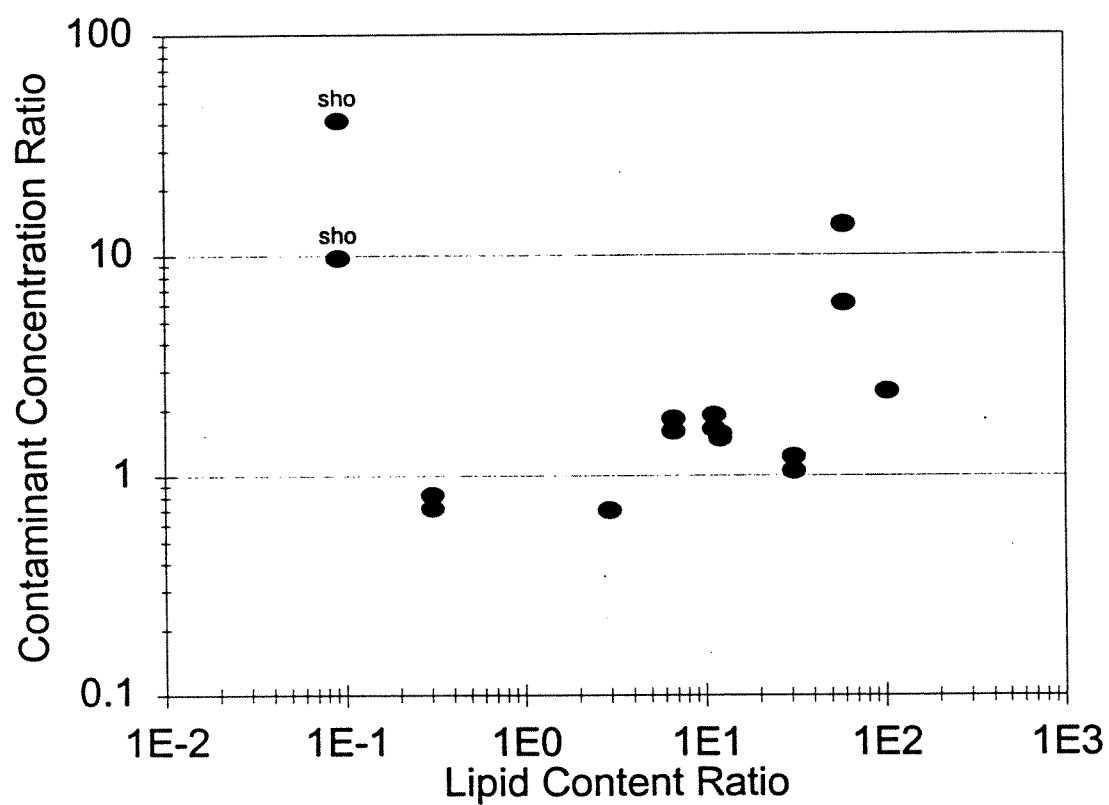
Contaminant concentrations on a lipid basis.

Figure 6-6.   The relationship between contaminant and lipid contents in liver and muscle
of fish. Horizontal axis: [g lipid/g wet weight in liver]/[g lipid/g wet weight
in muscle].

ratio appears to increase with the liver/muscle lipid content ratio (Figure 6-1).

A relationship between lipid content ratio and lipid-based contaminant level ratio would occur if the relative proportions of lipid types in the liver and muscle tissues covaried with the lipid content ratio. Two major classes of tissue lipids are polar and nonpolar. Membrane phospholipids are an example of polar lipids, and triglycerides that are used for energy storage are examples of nonpolar lipids. The method of extraction determines which class(es) of lipids are extracted. Some studies have shown that the solubility of PCBs in nonpolar lipid is much greater than in polar lipid (e.g., Schneider 1982), while others, using different extraction techniques, concluded that bound, or polar, lipids do accumulate significant quantities of PCBs (de Boer 1988).

The fillet and liver concentration of nonpolar lipid varies more than the concentration of polar lipids (Ewald and Larsson 1994, Scheider 1982). Thus, variation in the lipid-based liver/muscle contaminant ratio may be due to variation in the amount of nonpolar lipid in the muscle and in the liver tissue. Samples with a high nonpolar lipid content in the liver and a low nonpolar lipid content in the muscle are expected to have a higher lipid-based contaminant level in the liver, if contaminants are more soluble in nonpolar than in polar lipids. This is consistent with the pattern seen in Figure 6-6; at high liver/muscle lipid content ratio, the liver/muscle lipid-based contaminant ratio is high. A full quantitative understanding of the relationship must await more information concerning contaminant solubilities in polar and nonpolar lipids, as well as polar and nonpolar lipid concentrations in the two tissue types. For the present, the data-based relationship is used to establish a conservative estimate of the lipid-based liver/muscle contaminant ratio.

Species-specific liver/muscle conversion factors based on data collected in the Southern California Bight were used for the white croaker and kelp bass (Table 6-2). For the Dover sole a generic liver/muscle conversion factor was computed using the data collected in Table 6-2. Values for the shortspine thornyhead were considered outliers based on the relationship between contaminant ratio and the ratio of lipid contents (Figure 6-6), and therefore were not included in the analysis. The average lipid-based liver/muscle contaminant ratio was 2.61 (standard deviation = 3.45, n=14, data from eight species, for total PCB and p,p'DDE).

Muscle/Ovary Relationship

To develop a relationship between ovary and muscle lipid-based contaminant levels, total PCB and p,p'DDE concentrations measured in these two tissues in several species of fish were compiled from published literature. Only studies in which ovary and muscle were sampled from the same fish were used. In all cases, eggs or gonads were collected from ripe females or females in the process of developing eggs. These values include fatty and lean species, and iteroparous and semelparous species. The lipid-based concentration ratio calculated from the results of each study are given in Table 6-3. The units of these values are:

$$\frac{\mu g \text{ contaminant/g lipid in gonad}}{\mu g \text{ contaminant/g lipid in muscle}} \tag{6-3}$$

Also tabulated in Table 6-3 is the ratio of lipid contents in the two tissues. The units of these values are:

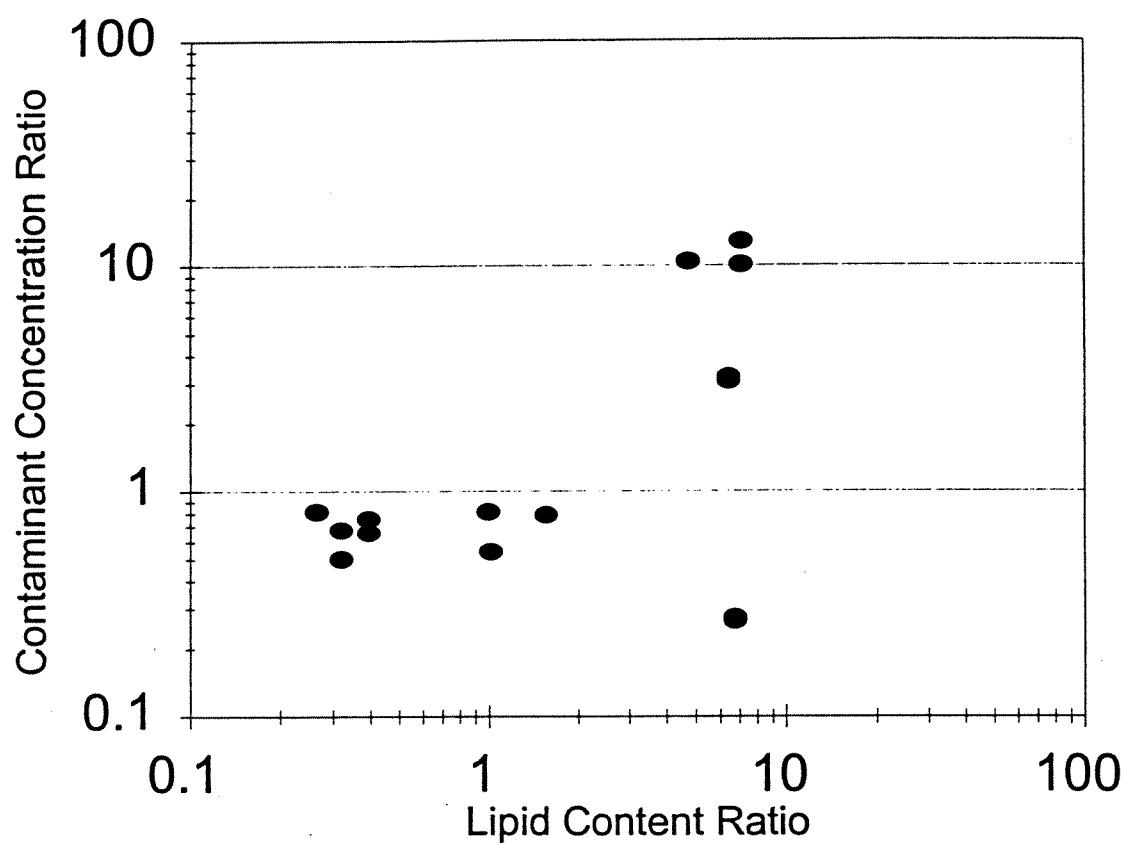$$\frac{g \text{ lipid/g wet weight in gonad}}{g \text{ lipid/g wet weight in muscle}} \tag{6-4}$$

Measured lipid-based ovary/muscle contaminant ratios vary over almost a factor of 50 (from 0.27 to 12.9; Table 2-2). To explore the causes for this variation, the relationship between this ratio and the ratio of lipid contents was studied (Figure 6-7). The ovary/muscle contaminant ratio is relatively constant up to a lipid ratio of approximately 2; as the lipid content ratio increases beyond 2, the contaminant ratio tends to increase. One point is inconsistent with this pattern; the point at 6.8, 0.27 (lipid ratio, contaminant ratio) is for spawning chinook salmon. This species is semelparous and expends a high proportion of its energy budget in the process of reproduction; its ratios may be sensitive to the time during the reproductive cycle when measurements are performed. Without the chinook salmon value, the pattern is qualitatively the same as that seen in the liver/muscle ratio data (Figure 6-6).

Ovary/muscle contaminant ratios are known for the white croaker; ovary/muscle ratios for the kelp bass and Dover sole must be estimated from literature-based values in Table 6-3 and Figure 6-7. To provide a conservative estimate of ovary concentrations, the data for those

Table 6-3. Ovary/Tissue Contaminant Ratios for Fish

Species	Location	Tissue	Chemical	Ratios		Reference
				Lipid content	Lipid-based Contaminant Concentration	
Baltic herring	German Baltic coast	muscle	DDE	0.321	0.675	Hansen et al. 85
Baltic herring	German Baltic coast	muscle	PCB 1260	0.321	0.501	Hansen et al. 85
chinook salmon	Great Lakes	muscle	PCB	6.800	0.275	Miller 93
chinook salmon	Great Lakes	muscle	p,p'DDE	6.800	0.265	Miller 93
dab	coastal France	muscle	PCB 138,153	1.021	0.539	Loizeau&Abarnou 94
eel	St.Lawrence estuary	carcass	PCB	1.568	0.786	Hodson et al. 94
eel	St.Lawrence estuary	carcass	p,p'DDE	1.000	0.811	Hodson et al. 94
lake trout	Lake Michigan	whole body	PCB congs	0.267	0.810	Mac et al. 93
lake trout	Great Lakes	muscle	PCB	0.398	0.656	Miller 93
lake trout	Great Lakes	muscle	p,p'DDE	0.398	0.753	Miller 93
striped bass	Nova Scotia river	muscle	PCB	4.745	10.455	Ray et al. 84
pike	Scandinavian lake	muscle	p,p'DDE	7.130	12.846	Larsson et al. 93
pike	Scandinavian lake	muscle	PCB	7.130	10.142	Larsson et al. 93
white croaker	SCB	muscle	tDDT	6.5	3.24	SCCWRP 86, Risebrough 87
white croaker	SCB	muscle	PCB	6.5	3.08	SCCWRP 86, Risebrough 87

Note: Units are given in the text.

Contaminant concenetrations on a lipid basis.
Two values are computed on a whole body, or carcass basis

Figure 6-7.   The relationship between contaminant and lipid contents in ovary and muscle
of fish. Horizontal axis: [g lipid/g wet weight in ovary] / [g lipid/g wet weight
in muscle].

6-16

species in Table 6-3 and Figure 6-7 with lipid content ratios less than 2 were used. This approach is conservative, because the ovary concentrations calculated from this ratio are less than or equal to those that would be calculated if the all of the data were used. The average lipid-based ovary/muscle contaminant ratio of these data is 0.69 (standard deviation = 0.12, n=8, data from four species, for total PCB and p,p'DDE).

Lipid-based ovary/muscle contaminant ratios for the white croaker were calculated by multiplying measured ovary/liver ratios by measured liver/muscle ratios. Lipid content ratios were calculated in a similar fashion. These calculations were based on matched liver and ovary data from Los Angeles Harbor and Dana Point (Southern California Coastal Water Research Project (SCCWRP) 1986) and on matched liver and muscle data from several sites in the Southern California Bight (Risebrough 1987; Table 6-2). The ovary/liver data consisted of 80 samples for total DDT and 68 samples for total PCB in which contaminants were measured in both the liver and the ovaries of the same fish. One outlier with a high value was removed from analysis (Figure 6-8). There was no significant difference between the samples collected in the two locations (Student's t test, $P >> 0.05$). The arithmetic mean ovary/liver contaminant ratios were 1.79 (standard deviation = 2.80, n=79) for total DDT and 1.94 (standard deviation = 2.25, n=67) for total PCB. The ovary/liver lipid concentration ratio was 0.95 (standard deviation = 0.76, n = 80). The calculated muscle/ovary lipid and contaminant ratios for white croaker are given on Figure 6-7 and in Table 6-3.

6.3.2 Potential Biases in the Calculation

It is necessary to account for two potential biases in the data. First, differences between the sexes in contaminant levels may affect results. The goal of this work is to calculate levels in gonads of females. However, the muscle and liver contaminant data from the Southern California Bight presumably include both males and females. Therefore, an analysis of male/female contaminant differences is performed using the white croaker data from the Southern California Bight, as well as published studies of male/female differences in contaminant levels in fish and published studies of the impact of egg production on contaminant loss rates in fish.

Second, seasonal variation in the relationships in contaminant levels among tissues may affect the relationship between the Southern California Bight data and calculated ovary values. There is seasonal variation in lipid content of fish tissues, in part related to reproduction. As
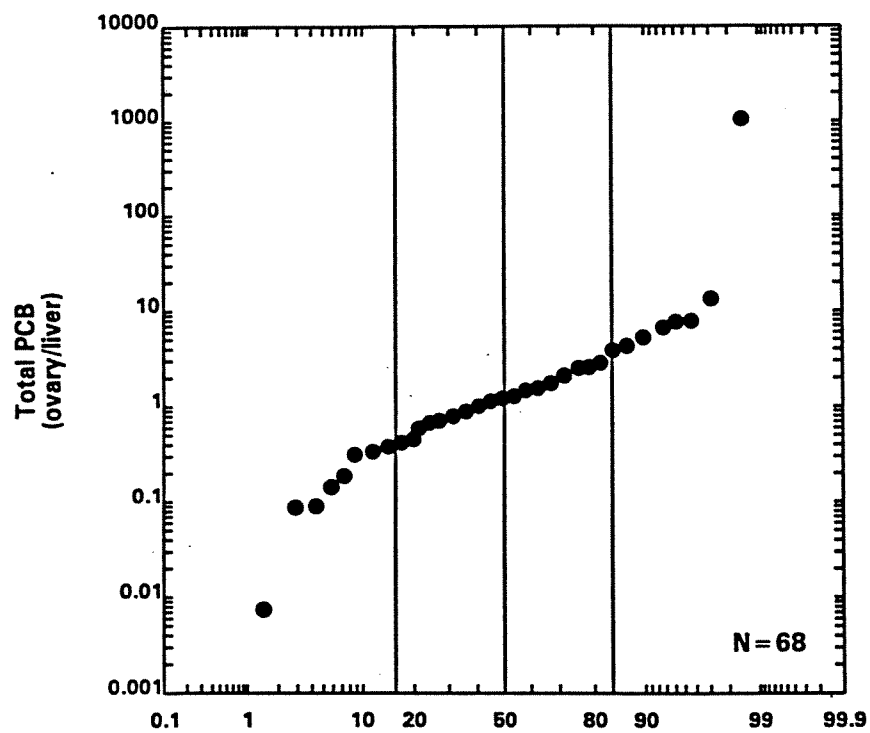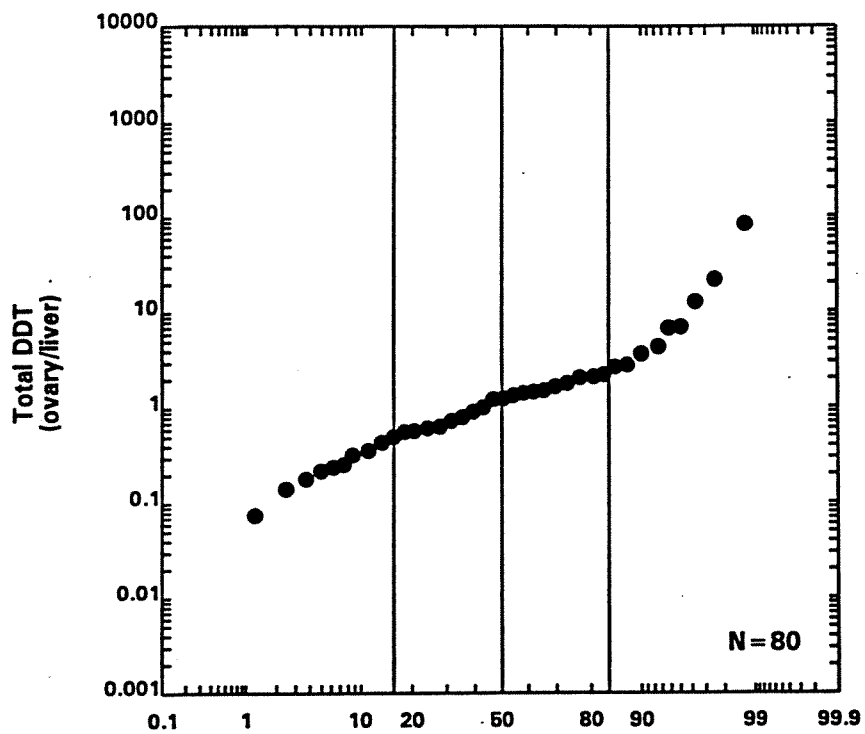
Figure 6-8. Ovary/liver contaminant ratios for white croaker collected in the SCB. Probability plot. Top panel: total DDT; Bottom panel: total PCB.

total lipid varies, the proportions of polar and nonpolar lipids within each tissue may vary, and this may lead to seasonal changes in the relative solubilities of contaminants in various tissues. Data from the Southern California Bight and from studies performed in other locations are used to explore the impact of this potential bias on the analysis.

Following the conversion of measured muscle and liver contaminant concentrations to equivalent ovary concentrations and consideration of variation due to agency/study, gender and season, the calculated lipid-based ovary concentrations are converted to wet weight-based concentrations using published estimates of ovary lipid contents. Then, the distribution of ovary contaminant concentrations (ppm wet weight) is plotted for each species for each HydroQual spatial segment. The proportion of each species in each spatial segment with calculated ovary contaminant levels greater than the specified concentration is then calculated.

Variation Between Males and Females

Differences in contaminant levels between males and females were studied in two ways: by comparing contaminant levels measured in both sexes in field populations and by calculating the loss of contaminant due to reproduction (sperm and eggs).

Data for total DDT and total PCB concentrations in white croaker collected in the Southern California Bight by Pollock (1991) were used to explore variation between sexes in lipid-based contaminant levels. All values in this data set were measured on composites of four fish, and each composite contained between zero and three females. A male/female difference in contaminant level should result in a non-zero slope when the lipid-based contaminant level in each composite and the number of females are plotted against each other. These samples were collected at a variety of locations, with differing contaminant levels. To remove these spatial differences from the analysis, the contaminant level in each composite was divided by the average contaminant level for all composites collected in the same location. There were no trends in the lipid-based contaminant levels as a function of the number of females in each composite (Figure 6-9).

Concentrations of total PCB were measured in male and female dabs (*Limanda limanda*) sampled from several locations in the North Sea (Knickmeyer and Steinhart 1989). The average ratio of female/male concentrations (ppm lipid) was 0.76 (95 percent confidence interval = 0.66, 0.86).
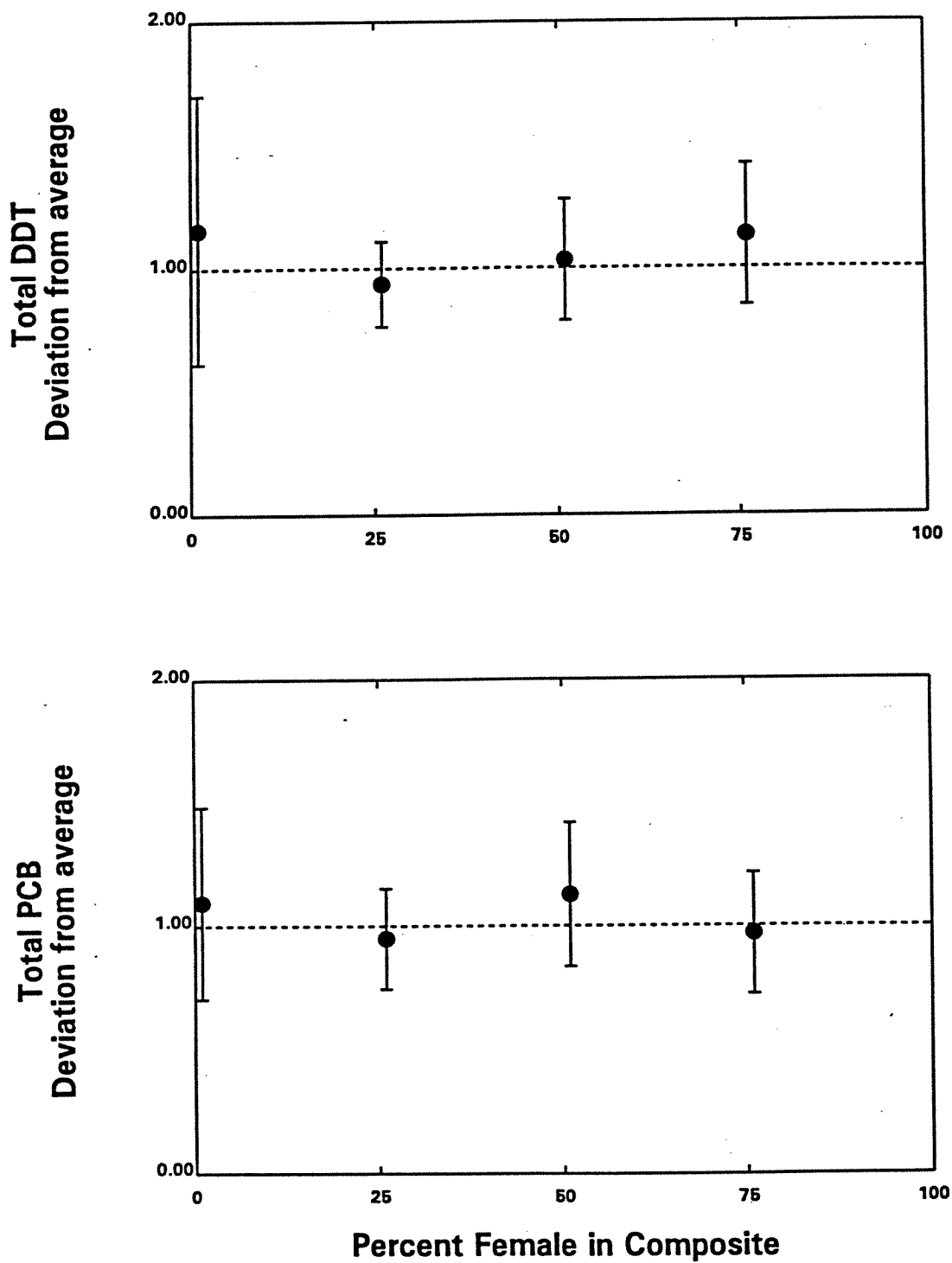
Figure 6-9. Effect of gender on contaminant levels. Relationship between the proportion of females in the composite and the lipid-based concentrations of total DDT and total PCB in muscle of white croaker. Data from Pollock (1991). Values are mean +/- two standard errors.

Concentrations of PCBs in whole bodies of male and female rainbow trout (*Salmo gairdneri*) were studied in fish that were exposed to radiolabeled PCBs in water for 36 hours and then placed in clean water for 52 weeks (Guiney *et al.* 1979). The half-lives of the decline of radiolabel during the non-breeding season were 1.76 years for females and 1.43 years for males. During the breeding season, the half-lives were 0.52 years for females and 0.54 years for males. Thus, loss of material to the eggs and sperm affects the annual average loss rate, but material appears to be lost in both females and males at approximately equal rates.

Thus, studies on contaminant levels in field populations of fish indicate that in some cases, no difference between females and males is discernible, and in some cases females have lower contaminant levels than males. A difference in contaminant levels between females and males would result in a biased estimate of ovary concentrations, because the data in the HydroQual database presumably include both males and females. The smaller female/male ratio estimated was 0.76. A conservative estimate of ovary tissue concentrations for the Southern California Bight would result if this smallest ratio is used; that is using 0.76 results in lower computed ovary concentrations. Therefore, female body burdens were assumed to average 25 percent lower than male body burdens, and the data in the HydroQual database were assumed to consist of equal numbers of males and females. The calculated concentrations were multiplied by 0.875 to estimate equivalent concentrations in females.

Variation Among Seasons

Two-thirds of the fish contaminant data records in the HydroQual database include an indication of month of collection. Data were collected in every month but December, with approximately 80 percent collected between May and September and 20 percent between October and February. Published contaminant levels in matched muscle and ovaries (discussed above) were measured in individuals collected during the spawning season. If lipid-based contaminant levels in muscle and liver vary during the year, then the data in the HydroQual database may provide a biased estimate of ovary levels.

Some measurements of seasonal variation in lipid-based contaminant levels have been made. For example, concentrations of PCBs were measured in brown trout, walleye, alewife, and rainbow smelt in Green Bay in spring, summer, and fall (Connolly *et al.* 1992). No consistent pattern of seasonal variation was apparent. Concentrations of PCBs and DDTs were measured in white croaker livers collected throughout 1985 in the Southern California Bight

(SCCWRP 1986). Lipid-based concentrations in liver varied over the year in females, being higher in fall and winter, when gonads are being formed, than during the spring and summer by a factor of 2 to 3.

The reproductive seasons of fish of the Southern California Bight vary. Winter and spring are the peak spawning period for species near the southern edge of their distribution, including, for example, Pacific hake and olive rockfish. Spring to summer are the peak spawning period for species near the northern edge of their distributions including, for example, kelp bass and queenfish (Cross and Allen 1994).

Thus, the relationship between lipid-based contaminant levels during the spawning season and other times of the year is not completely understood: the data from Green Bay indicated no consistent seasonal pattern, while the data for the white croaker in the Southern California Bight indicated higher lipid-based liver concentrations during the period of gonad formation. It is not clear whether in general the HydroQual database is biased, because the extent and pattern of seasonal variation in contaminant level is not known for each species, and because the timing of reproduction is variable between species. A bias might occur if data were not collected during the spawning season. Based on the white croaker data, such a bias would tend to be conservative, because lipid-based levels in livers were lower during the non-spawning season. That is, if there is a bias, the analysis based on the HydroQual database would lead to an underestimation of concentrations during the spawning season. Therefore, no correction is applied for seasonal variation in lipid-based contaminant concentration.

6.3.3 Ovary Lipid Content

The calculated lipid-based ovary contaminant concentrations must be converted to wet weight-based values. This conversion is based on the lipid content of the ovary. Lipid contents of eggs from several species of fish are presented in Figure 6-10 and Table 6-4. Values range from 0.013 g lipid/g wet weight for cod to 0.52 for striped bass. The median value is 0.08 g lipid/g wet weight. A more conservative value is one standard deviation below the mean, or 0.02 g lipid/g wet weight. This value is lower than 10 out of 11 data points; it is used in the calculations for the kelp bass and Dover sole. For white croaker, the average ovary lipid contents measured in Southern California Bight samples was used (0.041, average of 85 samples collected in 1985 and 46 samples collected in 1988) (SCCWRP 1986; Hose and Cross 1994).
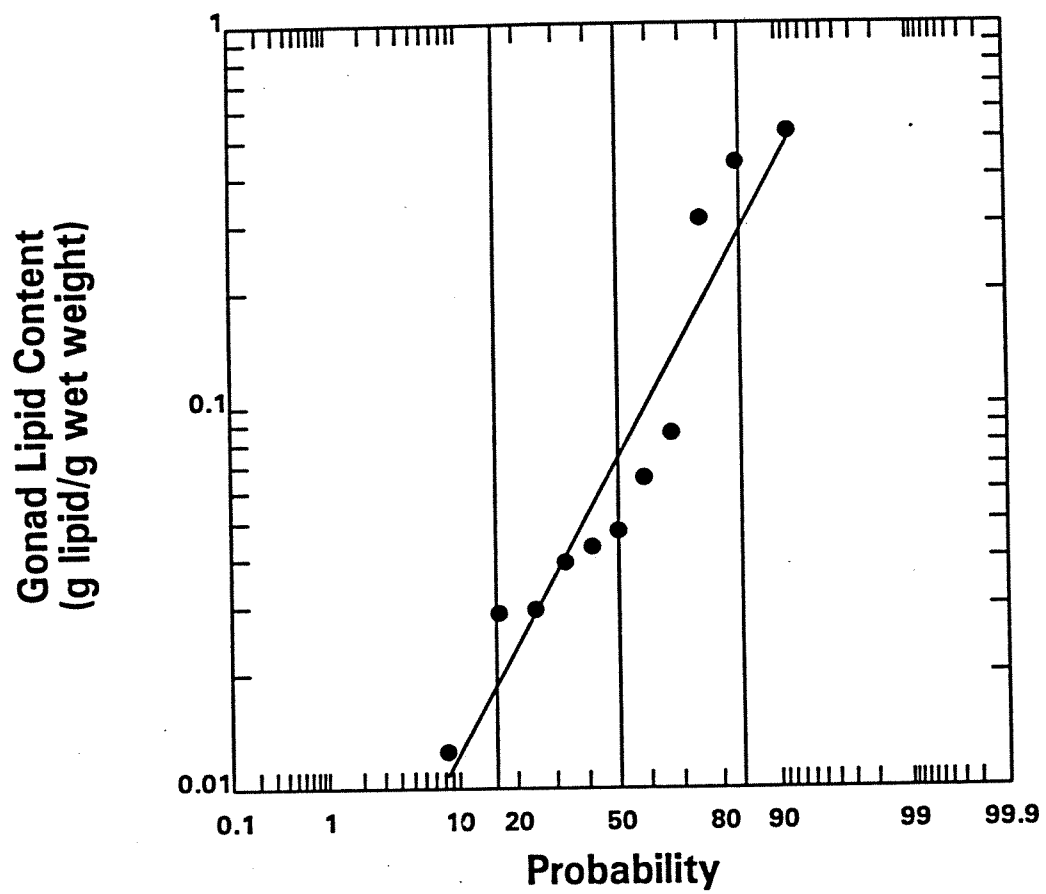
Figure 6-10. Distribution of gonad lipid contents (g lipid/g wet weight) for several species.

Table 6-4. Proportion Lipid in Fish Ovaries

Species	Proportion lipid	Reference
Baltic herring	0.029	Hansen *et al.* 85
chinook salmon	0.085	Miller 93
dab	0.029	Loizeau&Abarnou 94
eel	0.310	Hodson *et al.* 94
lake trout	0.047	Mac *et al.* 93
lake trout	0.065	Miller 93
cod	0.013	Schneider 82(5)
striped bass	0.522	Ray *et al.* 84
striped bass	0.433	Ray *et al.* 84
northern pike	0.039	Larsson *et al.* 93
white croaker	0.041	Hose&Cross 94, SCCWRP 86

6.3.4 Ovary Results

Calculations were performed for total PCB and total DDT. As in the analysis of muscle tissue, where only p,p'DDE data were available, these were divided by 0.87 to give equivalent total DDT values. All records in the HydroQual database that contained sufficient information (lipid content, etc.) were used. All calculations of ovary concentrations were performed on a lipid basis. The white croaker, kelp bass and Dover sole analyses were performed using the available species-specific conversion factors given previously. The following generic factors were used when species-specific values were not available:

- ovary concentration (ppm lipid) = muscle concentration (ppm lipid) X 0.69

- ovary concentration in females = calculated ovary concentration X 0.875 (to account for differences between males and females)

- ovary concentration:
  ppm wet weight = ppm lipid X 0.02 g lipid/g wet weight

Probability plots of ovary and calculated ovary total DDT levels for white croaker, kelp bass, and Dover sole collected in the Southern California Bight are presented in Appendix E. Bar charts of the results for these three species are presented in Figure 6-11.

The proportion of exceedances in each segment is related to the distance from the Whites Point Outfall, as expected based on the spatial patterns of average concentration presented in Section 2. White croaker exhibits a maximum of 100 percent exceedance in four segments. Dover sole exhibits a maximum of 45 percent, and kelp bass exhibits a maximum of 5 percent. This relationship (white croaker > Dover sole > kelp bass) is consistent with the relative average contaminant concentrations in each species (Section 2).
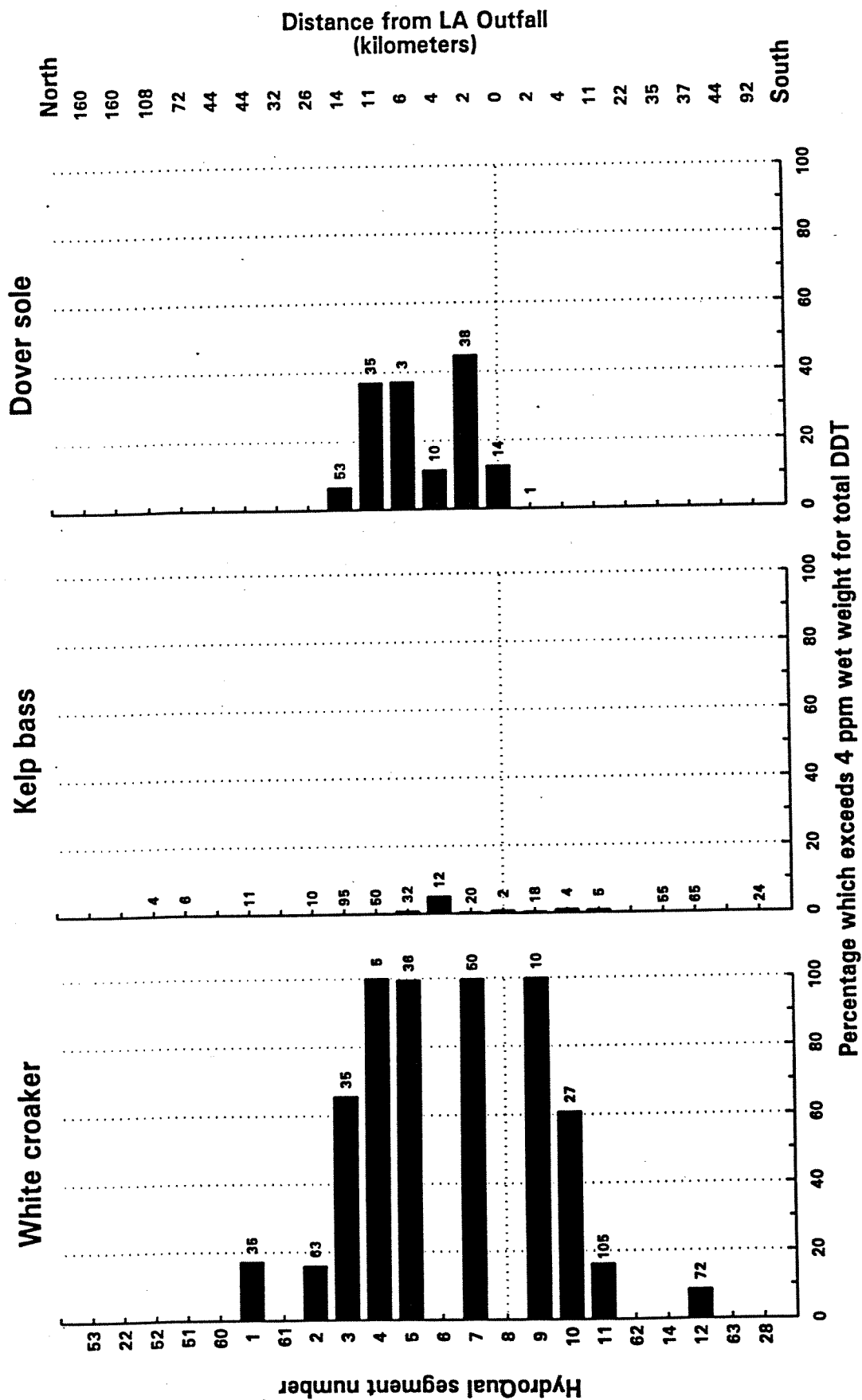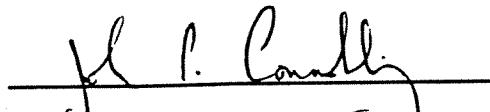
Figure 6-11. Computed percentage of fish with ovary or ovary equivalent total DDT concentration greater than 4 ppm wet weight. The number of samples per segment is posted to the right of each bar.
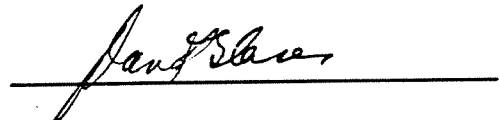
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The report was authored by John P. Connolly and David Glaser. Dr. Connolly directed the project and oversaw all its aspects. Technical responsibility for individual tasks was split between Dr.'s Connolly and Glaser. They shared responsibility for the data analysis and the development of the fish food web bioaccumulation models. Dr. Connolly had primary responsibility for the sea lion bioaccumulation model and Dr. Glaser had primary responsibility for the bird bioaccumulation models.

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SECTION 8

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APPENDIX A

BIOACCUMULATION MODELING THEORY

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BIOACCUMULATION MODELING THEORY

A.1   AQUATIC FOOD WEB

The accumulation of a contaminant by an aquatic animal includes the following processes:

- uptake and loss across the gill membrane
- uptake and loss across the gut wall
- hepatic and/or renal excretion
- non-hepatic metabolism
- growth dilution

The basic mass balance equation for an exposed animal defines the change in mass within the animal as being the difference between the above uptake and loss processes. In final form this equation is written as:

$$\frac{dv}{dt} = K_u c_d + \alpha C v_p - (K + G)v \qquad (A-1)$$

where:

v = concentration of contaminant in the animal (μg/g(w))

v_p = concentration of contaminant in prey (μg/g(w))

K_u = uptake rate from water (l/g(w)-d)

c_d = concentration of contaminant in water (μg/l)

α = assimilation efficiency of contaminant in food

C = consumption rate of food (g(w)/g(w)-d)

K = total excretion rate (1/d)

G = growth rate of the animal (g(w)/g(w)-d)

The first term of Equation (A-1) represents the direct uptake of contaminant by the animal from water. The second term represents the flux of contaminant into the animal through feeding. The third term is the loss of contaminant due to desorption and excretion plus the change in concentration due to growth.

Equation (A-1) is applied to each of the animals that comprise the food web. For the upper levels of the food chain changes in contaminant concentration with age are sometimes significant and each year class of the species at these levels is modeled separately.

The equations used to describe the individual processes within Equation A-1 have been presented in detail elsewhere (Connolly, 1991; Connolly *et al.* 1992) and will be summarized here.

A.1.1 Contaminant Mass Transfer at the Gill

The contaminant uptake rate constant K_u is defined from a mass transfer coefficient k_{gl} and the active gill surface area A_{gl}. However, it is not necessary to explicitly define these parameters. Rather, K_u may be determined from the oxygen uptake rate constant K_{uO2} and the ratio of the mass transfer coefficients of the contaminant and oxygen. Oxygen transfer rate is defined by the respiration rate of the animal and the oxygen concentration of the water $(c_{O2}, gO_2/l)$:

$$K_{uO_2} = \frac{r}{c_{O_2}} \qquad (A-2)$$

where r is the respiration rate in units $gO_2/g(w)$-d. Mechanistically this uptake rate may be described in term of a mass transfer rate constant at the gill (k_{glO2}).

$$K_{uO_2} = \frac{k_{glO_2} A_{gl}}{w} \qquad (A-3)$$

where w is the wet weight of the animal $(g(w))$.

Equations (A-2) and (A-3) may be equated and solved for A_{gl}. Substituting this expression for A_{gl} in the equation describing K_u yields:

$$K_u = \frac{k_{gl}}{k_{glO_2}} \frac{r}{c_{O_2}} \qquad (A-4)$$

The gill elimination rate, K_l, is given as:

$$K_l = \frac{K_u \rho_a}{x_a + \pi_L x_L} \tag{A-5}$$

where:

ρ_a = density of aqueous blood (g/l)

x_L = fraction lipid of the animal (g lipid/g(w))

x_a = $1 - x_L$

π_L = equilibrium partition coefficient of the contaminant between the lipid and aqueous phases of the animal

For most organic contaminants it appears that the gill is the major site of depuration (Gobas *et al.* 1989) and therefore K_l is equivalent to the whole-body loss rate K.

A.1.2 Contaminant Mass Transfer at the Gut Wall

The contaminant assimilation efficiency, α, is the ratio of the transfer rate from the gut to the animal to the total transfer rate out of the gut. Because these transfer rates are difficult to define, empirical estimates of α are used directly in the model.

Analysis of laboratory data for chemicals with log kows between 4.5 and 8 indicates that elimination across the gut is of limited importance relative to overall elimination rate (Connolly *et al.* 1992). Gobas *et al.* (1989) presented data that showed fecal elimination rate to be below gill elimination rate until log kow was above about 7. In any event, the fecal elimination rate is much less than the growth rate G and is not included in the model.

A.1.3 Consumption (Ingestion) Rate

The rate of consumption of prey C is calculated from the rate of energy usage by the animal. Energy usage is estimated from the rates of production and metabolism of body tissue by the animal. Growth rate defines the net production of body tissue (g(w)/g(w)/d). The rate of metabolism of body tissue, R, may be computed from the respiration rate, r, by: (1) stoichiometrically converting respiration from $gO_2/g(w)/d$ to $gC/g(w)/d$; (2) converting carbon to dry weight by assuming all animals are 40 percent carbon on a dry weight basis; and (3)

converting dry weight to wet weight using observed ratios. Given the caloric density of the animal's tissue in units cal/g(w), λ, the energy usage rate, P, is then;

$$P = \lambda(R + G) \tag{A-6}$$

Dividing P by the fraction of ingested energy that is assimilated, a, yields the rate of energy intake by the animal. The rate of consumption of prey, C, is the energy intake rate divided by the caloric density of the prey, λ_p.

$$C = \frac{\lambda}{\lambda_p} \frac{R + G}{\alpha} \tag{A-7}$$

Caloric density is computed from the composition of the animal and the caloric densities of lipid (39.5 kJ/g) and protein (20 kJ/g):

$$\lambda = 39.5 f_L + 20 f_{pr} \tag{A-8}$$

where:

f_{pr} = fraction protein = $f_L - f_d$
f_d = fraction dry

For deposit feeding animals the energy density of the prey (sediment) is specified on a carbon basis (i.e., kJ/gC). In the application of Equation A-1 v_p, the contaminant concentration in the prey (sediment), is expressed as $\mu g/gC$.

Ingestion of water provides an additional source of contaminant to the gut. The ingestion of contaminant by way of water ingestion is the product of a drinking rate, D (l/g(w)-d), and the total contaminant concentration in the water, c_T. This contaminant source is insignificant for two reasons. First, the contaminant concentration is orders of magnitude higher in the prey than in the water. To illustrate this point, consider a situation in which the predator ingests 0.01 g prey/g(w)-d (a low value representative of a 5 to 10 kg fish) and the partition coefficient (i.e., the bioaccumulation factor) between the prey and the water is 10^5 l/kg(w) (a reasonable value for PCBs in a lean forage fish). The drinking rate required to make ingestion from water equal to ingestion from food is given as follows.

$$D = N_p C \qquad \text{(A-9)}$$

where

N_p = prey bioaccumulation factor (l/kg(w))

Thus, the fish would have to drink 1,000 l/kg(w)-d or 5,000 to 10,000 l/d! Drinking is also insignificant in comparison to uptake from water. The uptake rate across the gill, K_u, has the same units as D and is directly comparable to D. Typical values of K_u range between 100 and 1000 l/g(w)-d. Again, the fish would have to drink incredible quantities of water to achieve an uptake equivalent to that obtained across the gill. For these reasons, ingestion of water is ignored in the models.

A.2 FEMALE SEA LIONS

The model for sea lions is derived from the model presented in A.1 but differs in that it lacks uptake and loss across a gill membrane and it includes transfer from mother to fetus and transfer from mother and nursing pup by lactation.

A.2.1 Contaminant Mass Transfer to the Fetus

The model assumes that the contaminant concentrations are equilibrated between the mother and the developing fetus. In other words, the contaminant concentrations in the lipid fraction of the mother and fetus are equal, as are the concentrations in the non-lipid or aqueous fraction. The concentration in the fetus is given as follows.

$$v_f = \epsilon v \qquad \text{(A-10)}$$

where

$$\epsilon = \frac{x_{a_f} + x_{L_f} \pi_L}{x_a + x_L \pi_L} \qquad \text{(A-11)}$$

the subscript f designates the fetus and terms are as defined previously. At birth the concentration in the mother changes due to the loss of the fetus. An increase or decrease may occur depending on the relative lipid contents of the mother and fetus. Concentration will

increase if x_{Lf} is lower than x_L and it will decrease if x_{Lf} is greater than x_L. Because the newborn sea lion has a lipid content of about 5 percent and the mother is about 30 percent lipid a concentration increase will occur. The post-birth:pre-birth ratio of concentrations is:

$$1 + \frac{w_f}{w}(1 - \epsilon) \qquad (A\text{-}12)$$

where

w_f = weight of the fetus

w = weight of the mother

A.2.2 Contaminant Mass Transfer By Lactation

The loss of contaminant by a lactating female is the product of a milk production rate, M (g milk/d), and the contaminant concentration in the milk, ν_m, divided by the weight of the female. The contaminant concentration in the milk is computed assuming that the milk lipids are in equilibrium with the whole-body aqueous or non-lipid fraction at a partition coefficient, π_m. Under this assumption the concentration in the milk is computed from the whole-body concentration by the following equation.

$$v_m = \left[\frac{1 + \pi_m x_{L_m}}{1 + \pi_L x_L}\right]v \qquad (A\text{-}13)$$

where

x_{Lm} = lipid fraction of the milk

The pups are assumed to assimilate all of the contaminant in the milk they ingest because nearly all of the lipids in the milk are digested by the pup. In addition to this exposure, pups that begin to forage also ingest contaminant from their prey. The ingestion rate for prey is determined from the difference between their energy usage rate (see A.1.3) and the energy provided by lactation.

A.3   BIRDS

The model for birds is derived from the model presented in A.2, but differs in that it lacks lactation and includes a dietary contribution to egg contaminant levels.   Contaminant concentration in the egg on a lipid basis (ug/g lipid) is equal to a weighted average of concentrations in dietary and body lipid:

$$v_e = \left[(1-P_e)\left(\frac{v}{x_L}\right) + P_e\left(\frac{v_p}{x_{L_p}}\right) \right] x_{Le} \qquad \text{(A-14)}$$

where:

v_e = concentration of contaminant in the egg (ppm wet weight)

v_p = concentration of contaminant in the prey (ppm wet weight)

v = concentration of contaminant in the animal (ppm wet weight)

P_e = proportion of dietary lipid in the egg

x_L = fraction lipid of the animal (g lipid/g(w))

x_{Lp} = fraction lipid of the prey (g lipid/g(w))

x_{Le} = fraction lipid of the egg (g lipid/g(w))

During the period of egg growth, the mother's weight increases and fraction lipid changes to account for the egg.   At laying, the weight, fraction lipid and contaminant concentration in the mother change due to loss of the egg.   Because the fraction lipid of the eggs is generally less than that of the mothers in the species studied here, at laying the mother's weight decreases, fraction lipid increases, and contaminant concentration increases on a wet weight-basis.

The total loss rate due to egg production, as an annual average rate is:

$$\text{LOSS} = \frac{v_{egg} \times W_{eggs} \times N_{eggs} \times N_{clutches}}{v_{female} \times W_{female}} \div 365 \qquad \text{(A-15)}$$

in which:

LOSS   =   loss rate (day^{-1})

v_{egg} =   concentration of the contaminant in the eggs (ppm wet)

W_{egg} =   weight of each egg (gwet/egg)

N_{egg} = number of eggs produced in each clutch

N_{clutch} = number of clutches per female produced each year

v_{female} = concentration of the contaminant in the female (ppm wet)

W_{female} = weight of female (gwet/female)

APPENDIX B

BOX PLOTS FOR FISH, MUSSEL AND SEDIMENT DATA ANALYSES

BOX PLOTS FOR FISH, MUSSEL AND SEDIMENT DATA ANALYSES

As described in Section 2.1, two methods of data presentation were employed. Data were presented as arithmetic means +/- 2 standards errors of the mean in Section 2. The same data are presented as Tukey box plots in Appendices B and C. To facilitate the comparison between the two presentation methods, figure numbers and titles remain the same except for the addition of the appendix number (i.e. Figure 2-1 is presented in this section as Figure B2-1).

Box plots are valuable analysis tools because no assumptions are made concerning the shape of the distribution of the data. The central tendency of the data is represented by the median and the spread of the actual data is indicated by boxes, whiskers and symbols. The edges of the central box, called hinges, represent the 25th and 75th percentiles of the distribution (i.e. 50 percent of the distribution falls within the box). To understand the remainder of the box plot, three terms need to be defined: the interquartile range (Hspread), the inner fence and the outer fence. The Hspread is the absolute value of the difference between the hinges. Inner fences are:

Lower inner fence = lower hinge - (1.5 * (median - lower hinge))
Upper inner fence = upper hinge + (1.5 * (upper hinge - median))

The outer fences are defined as:

Lower outer fence = lower hinge - (3 * (median - lower hinge))
Upper outer fence = upper hinge + (3 * (upper hinge - median))

Thus, the distance between the inner fences equals 2.5 * Hspread. The whiskers represent the range of values that fall within the inner fences. They do not necessarily extend to the inner fences; whiskers will only extend to the last real data point within the inner fences. Values between the inner fences and the outer fences are termed inner outliers, and are represented by asterisks. Values outside of the outer fences are called outer outliers and are plotted as open circles. In all plots with a logarithmic axis, the fences are computed using the logarithms of the data.
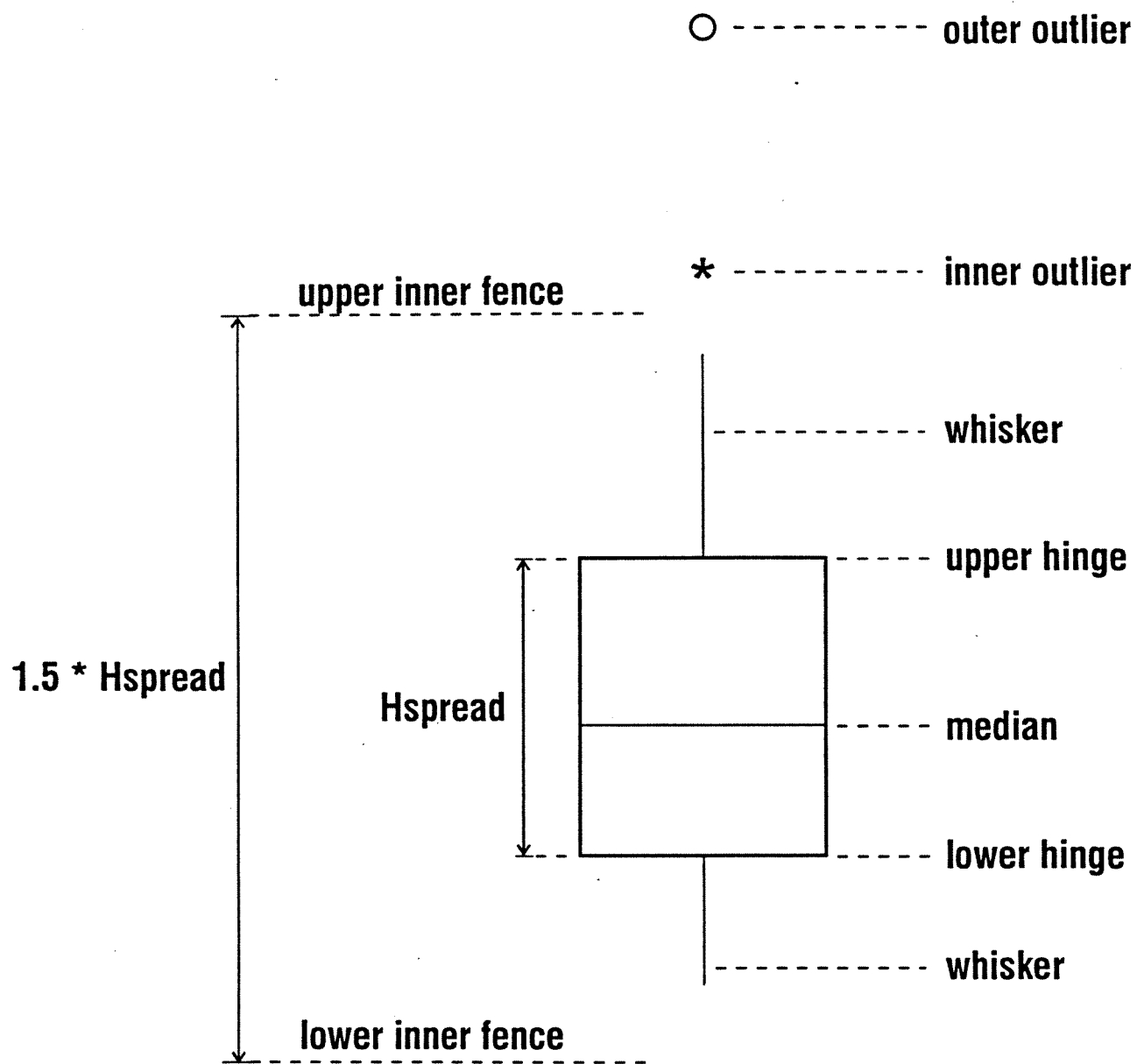
Figure B-1. Example of Box and Whisker Plot
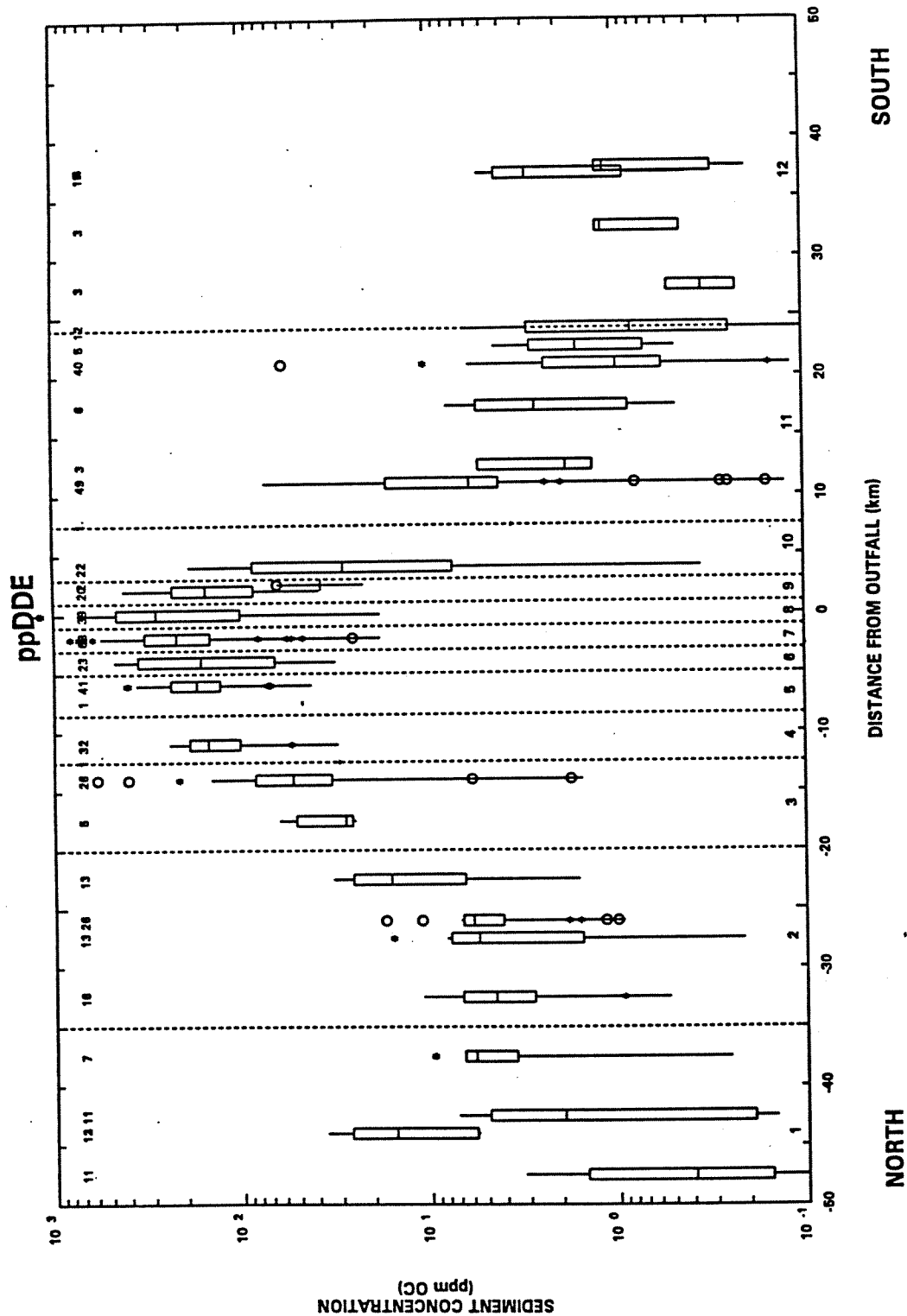
Figure B2-1. Observed p,p′DDE concentrations in surface sediments collected from the Southern California Bight between 1985 and 1995. Vertical dashed lines designate HydroQual segment boundaries. Segment numbers (1-12) are indicated at the bottom of the figure and sample sizes are indicated at the top. Concentrations are expressed as ppm organic carbon and are plotted as a function of distance from the Los Angeles County Outfall (segment 8; kilometer 0).
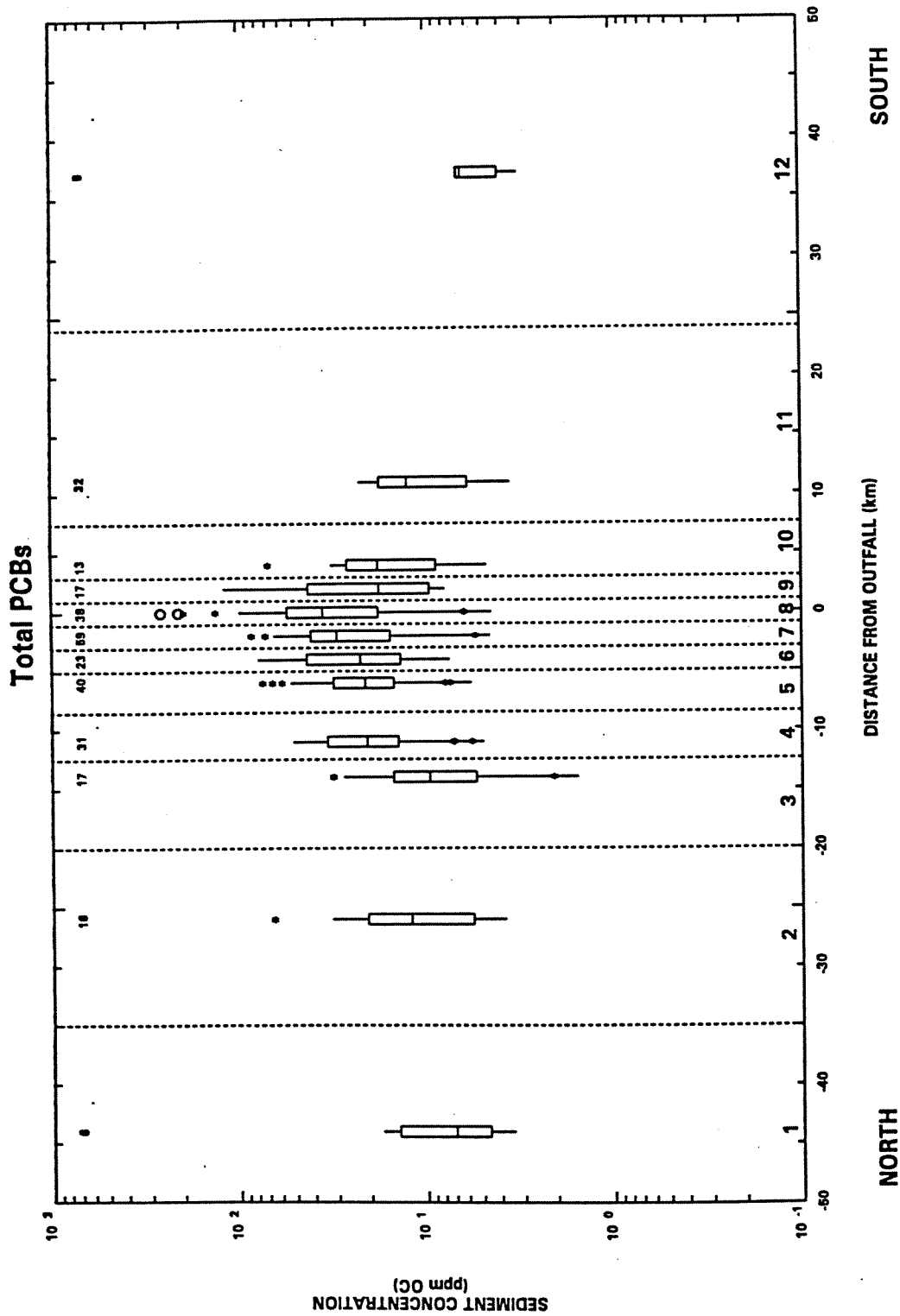
Figure B2-2. Observed total PCB concentrations in surface sediments collected from the Southern California Bight between 1985 and 1995. Vertical dashed lines designate HydroQual segment boundaries. Segment numbers (1-12) are indicated at the top. Concentrations are expressed as ppm organic carbon and are plotted as a function of distance from the Los Angeles County Outfall (segment 8; kilometer 0).
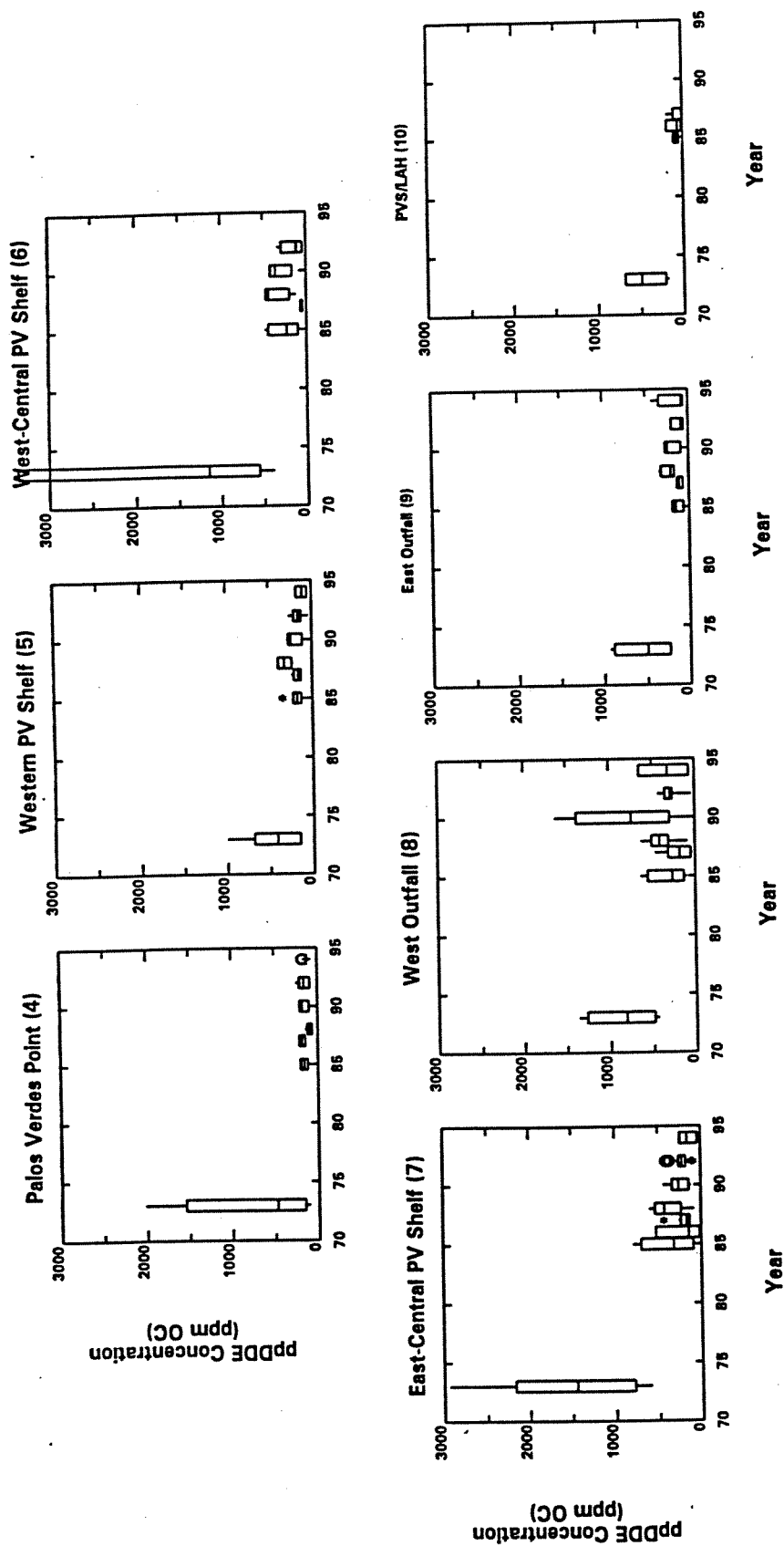
Figure B2-6. Temporal profiles of p,p'DDE concentrations in surface sediments for various segments of the Palos Verdes Shelf (ppm organic carbon).
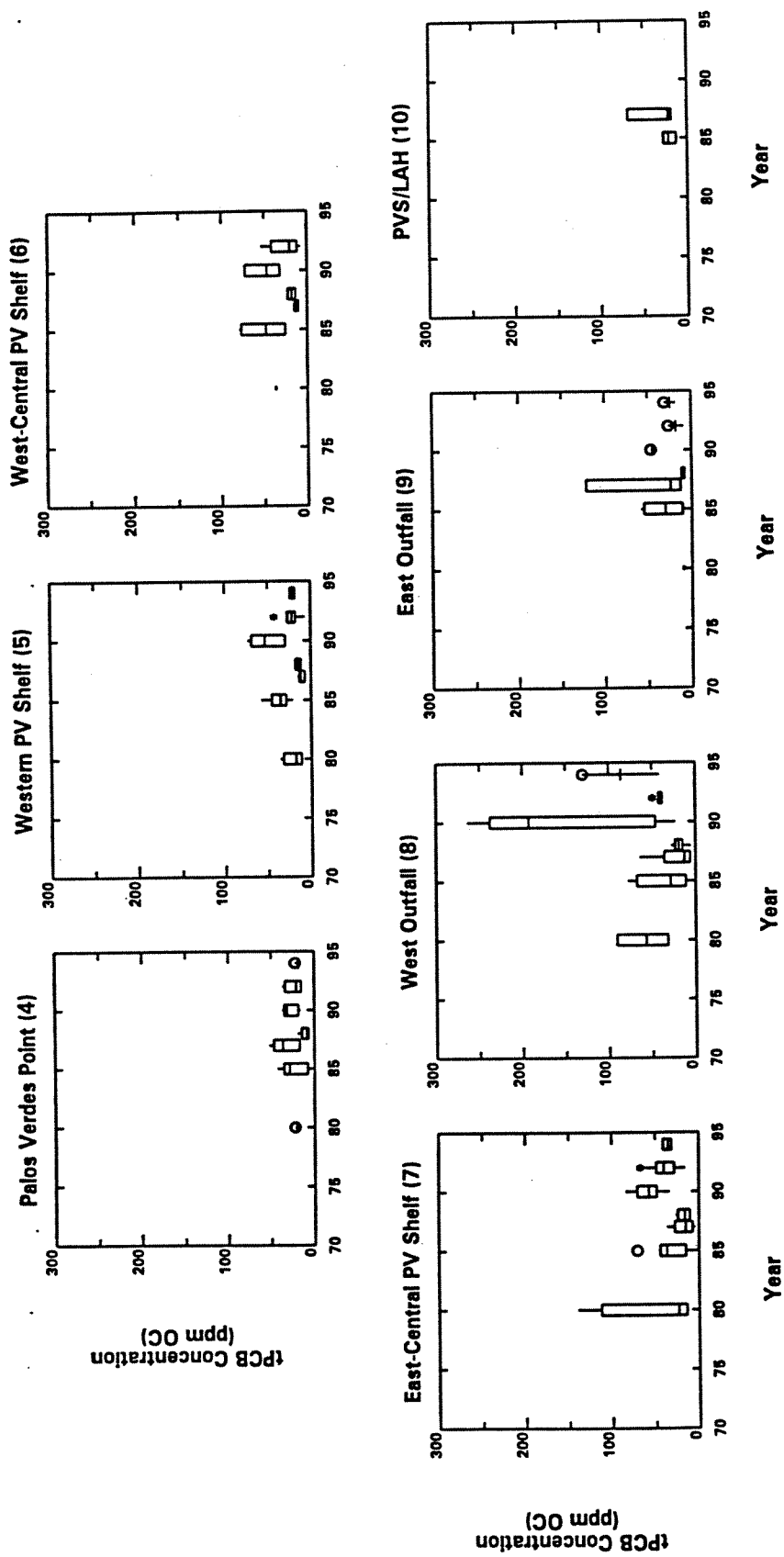
Figure B2-7. Temporal profiles of total PCB concentrations in surface sediments for various segments of the Palos Verdes Shelf (ppm organic carbon).
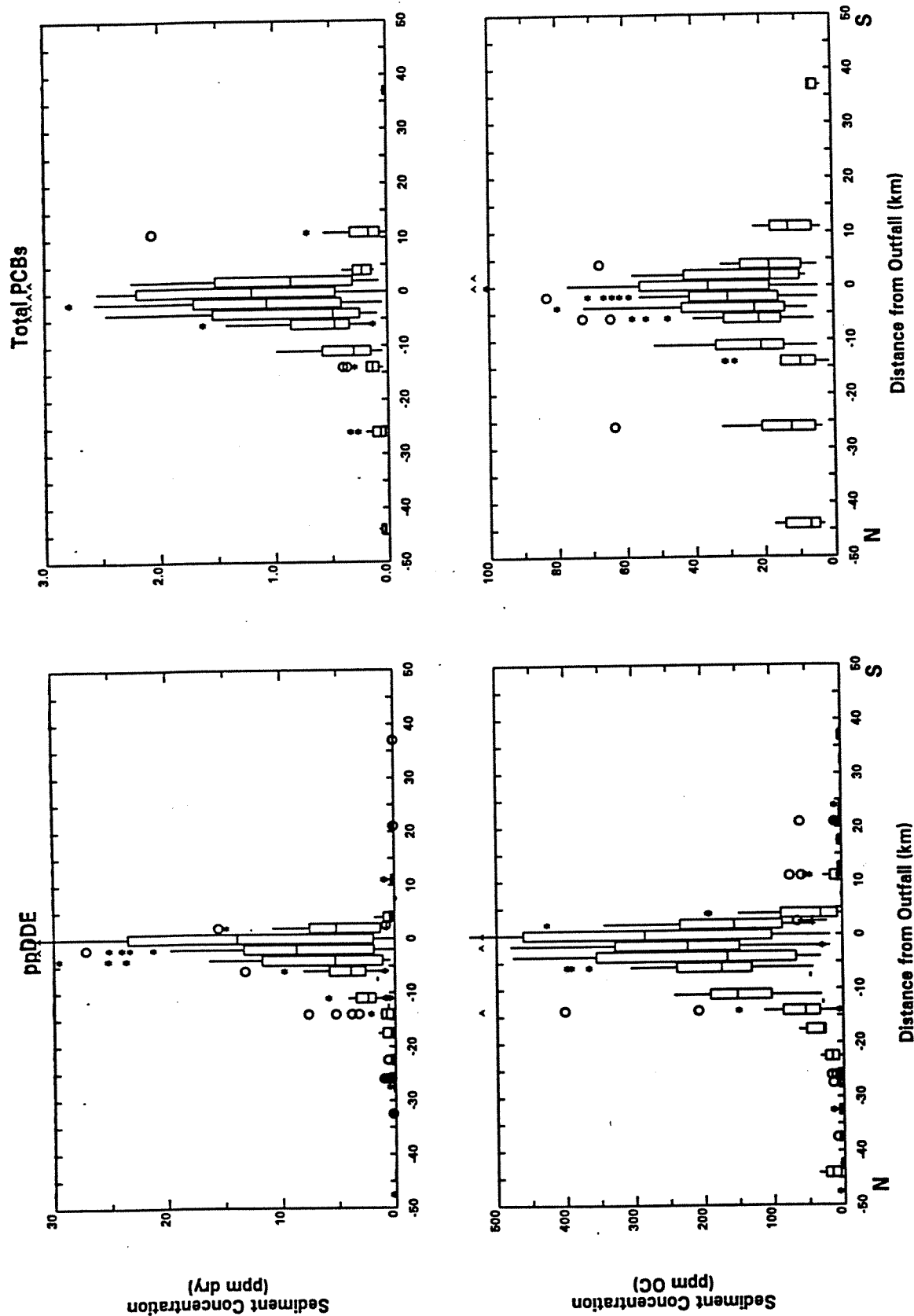
Figure B2-8. Spatial profiles of p,p'DDE and Total PCB levels in surface sediments collected from the Southern California Bight between 1985 and 1995. Top panels show data expressed as ppm dry weight. Bottom panels show data expressed as ppm organic carbon. Zero km point at Whites Point Outfall. Distances to the north are negative.
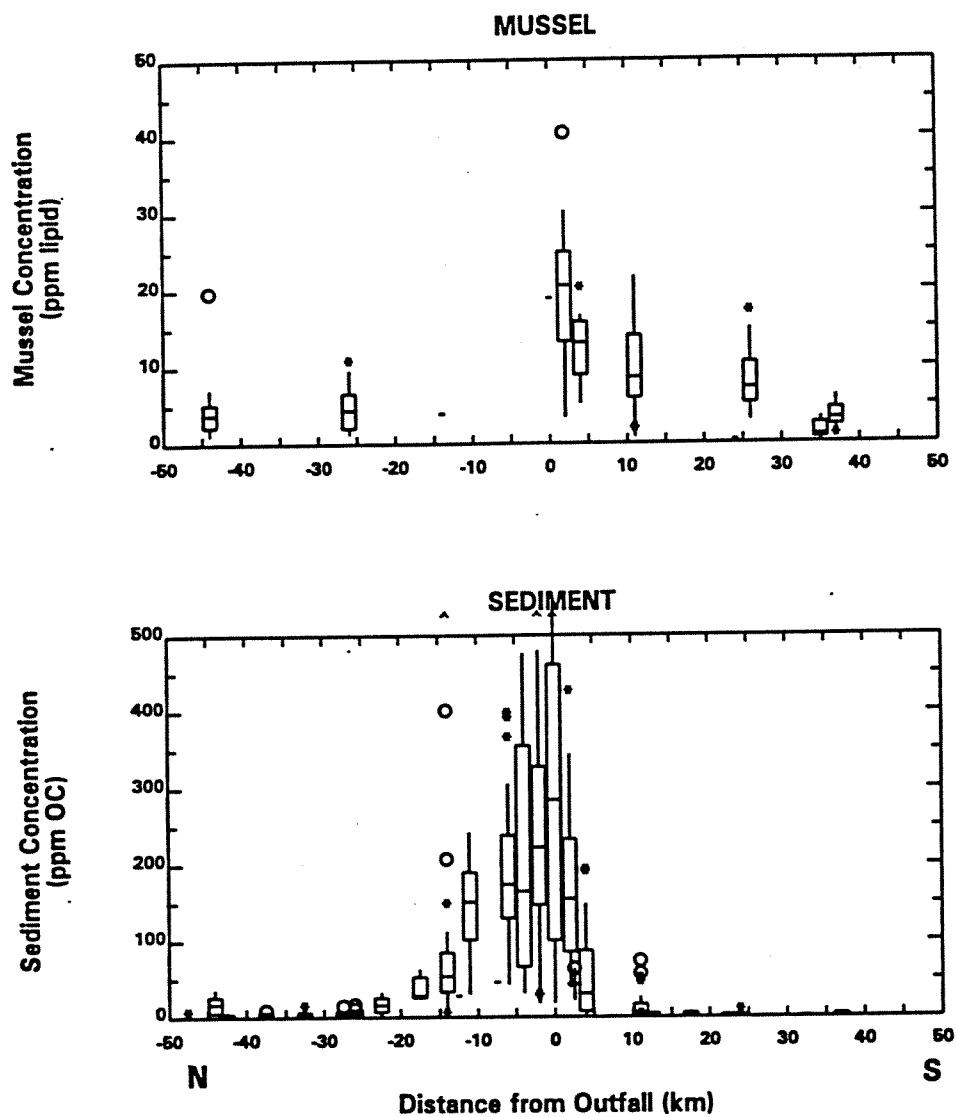
Figure B2-9. Spatial profiles of p,p'DDE concentrations in mussels and surface sediments collected from the Southern California Bight between 1985 and 1995. (ppm lipid, organic carbon. Zero km point at Whites Point Outfall. Distances to the north are negative.
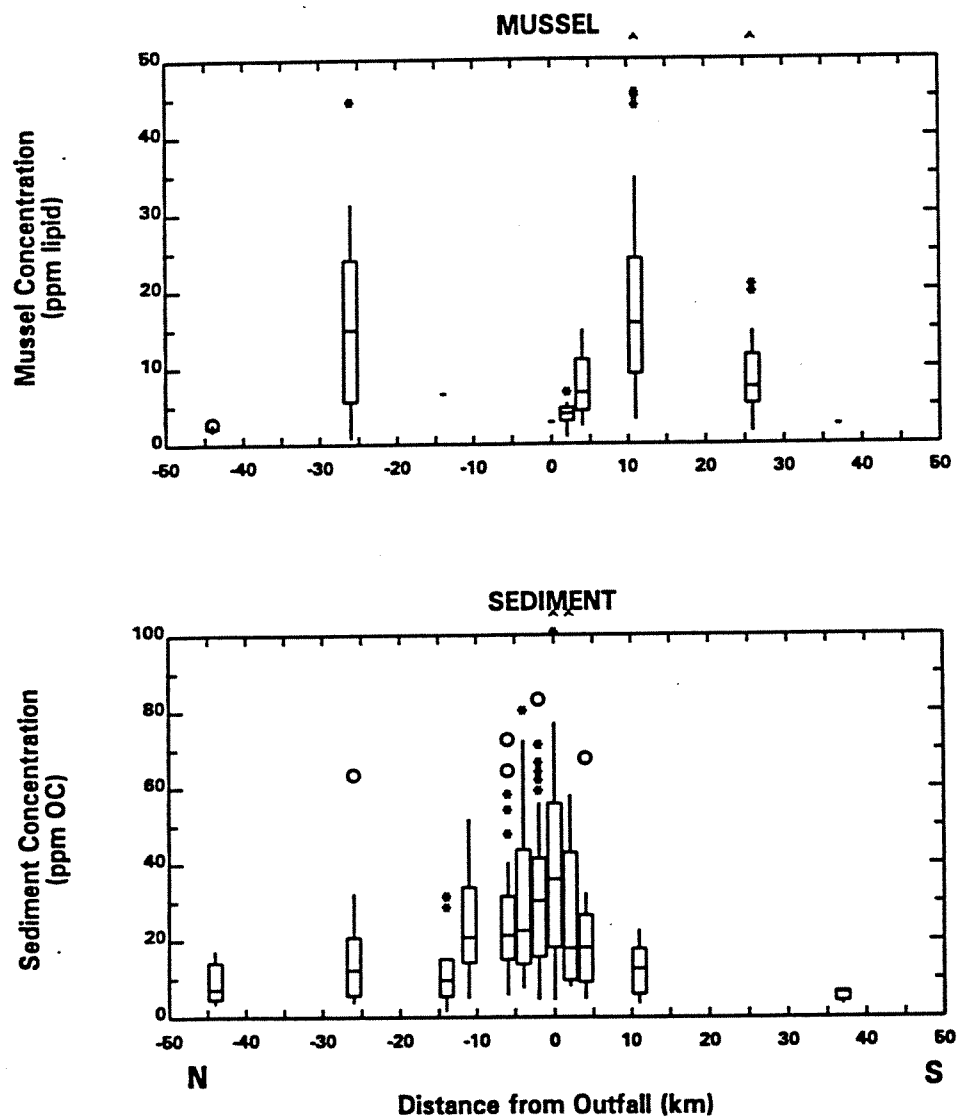
Figure B2-10. Spatial profiles of total PCB concentrations in mussels and surface sediments collected from the Southern California Bight between 1985 and 1995 (ppm lipid, organic carbon).
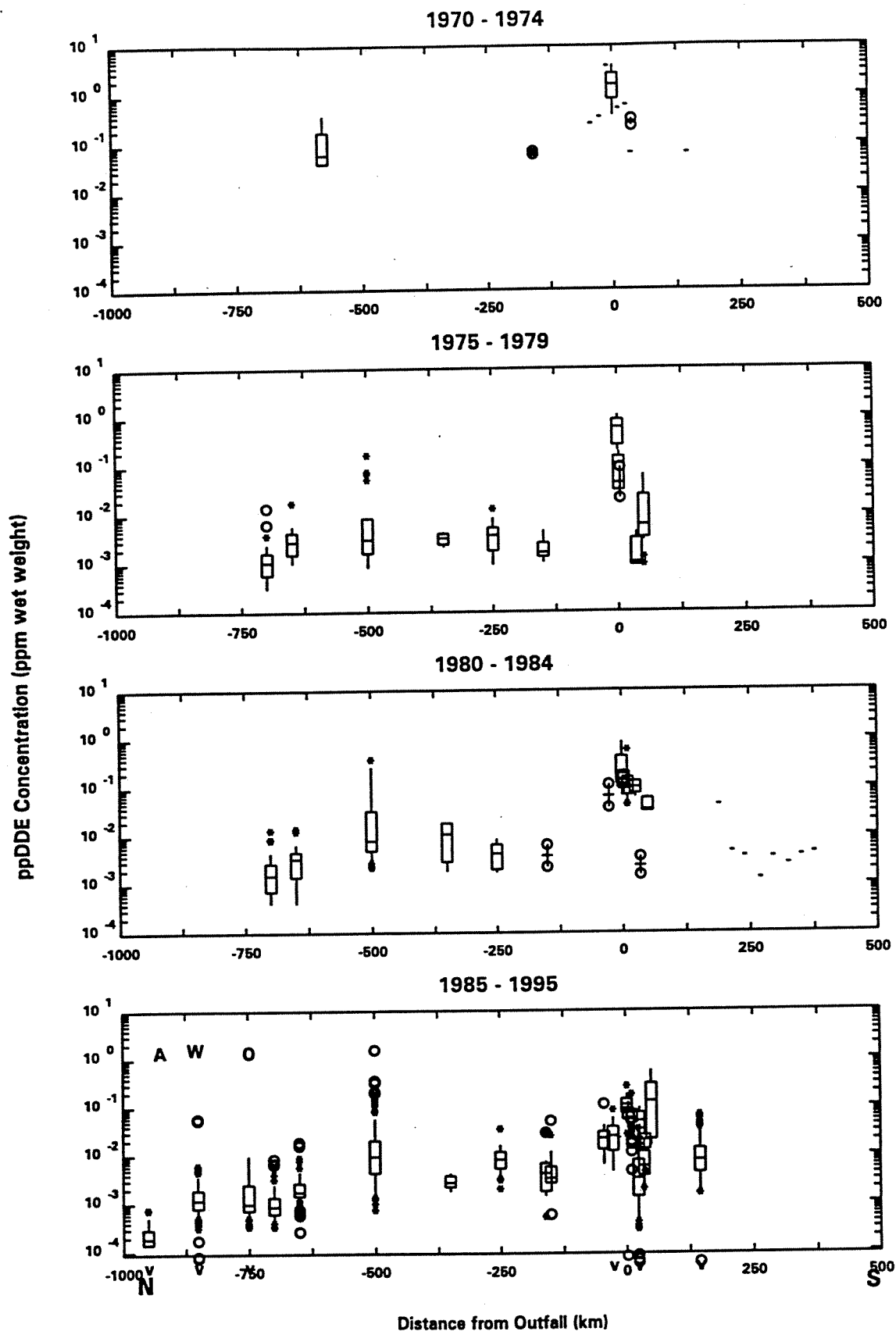
Figure B2-11. Spatial profiles of mussel p,p'DDE concentrations along the west coast between Alaska and Mexico for four time periods. (ppm wet weight). Zero km point at Whites Point Outfall. Distances to the north are negative (A-Alaska, W-Washington, O-Oregon).
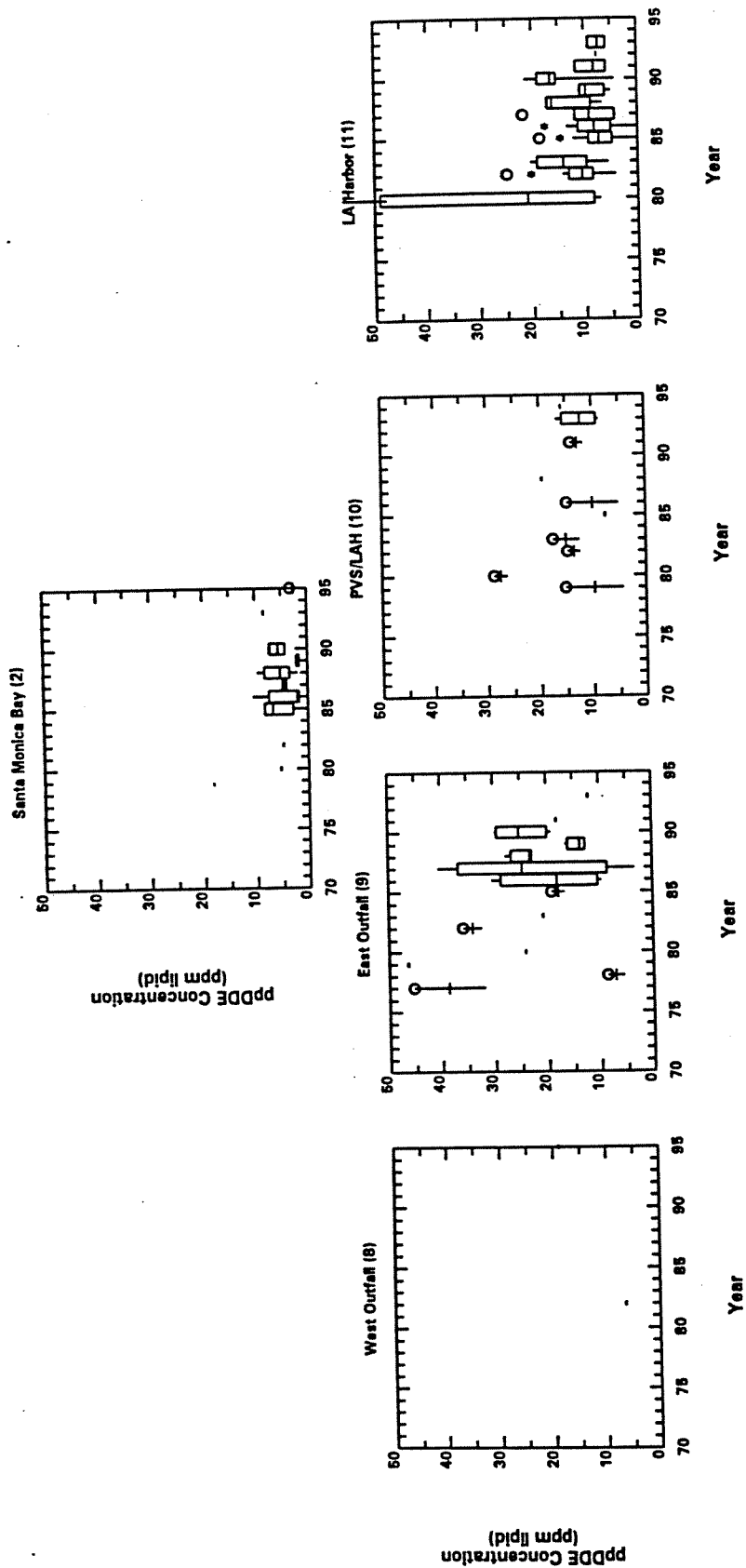
Figure B2-12. Temporal profiles of p,p'DDE concentrations in mussels for selected segments of the Southern California Bight. (ppm lipid).
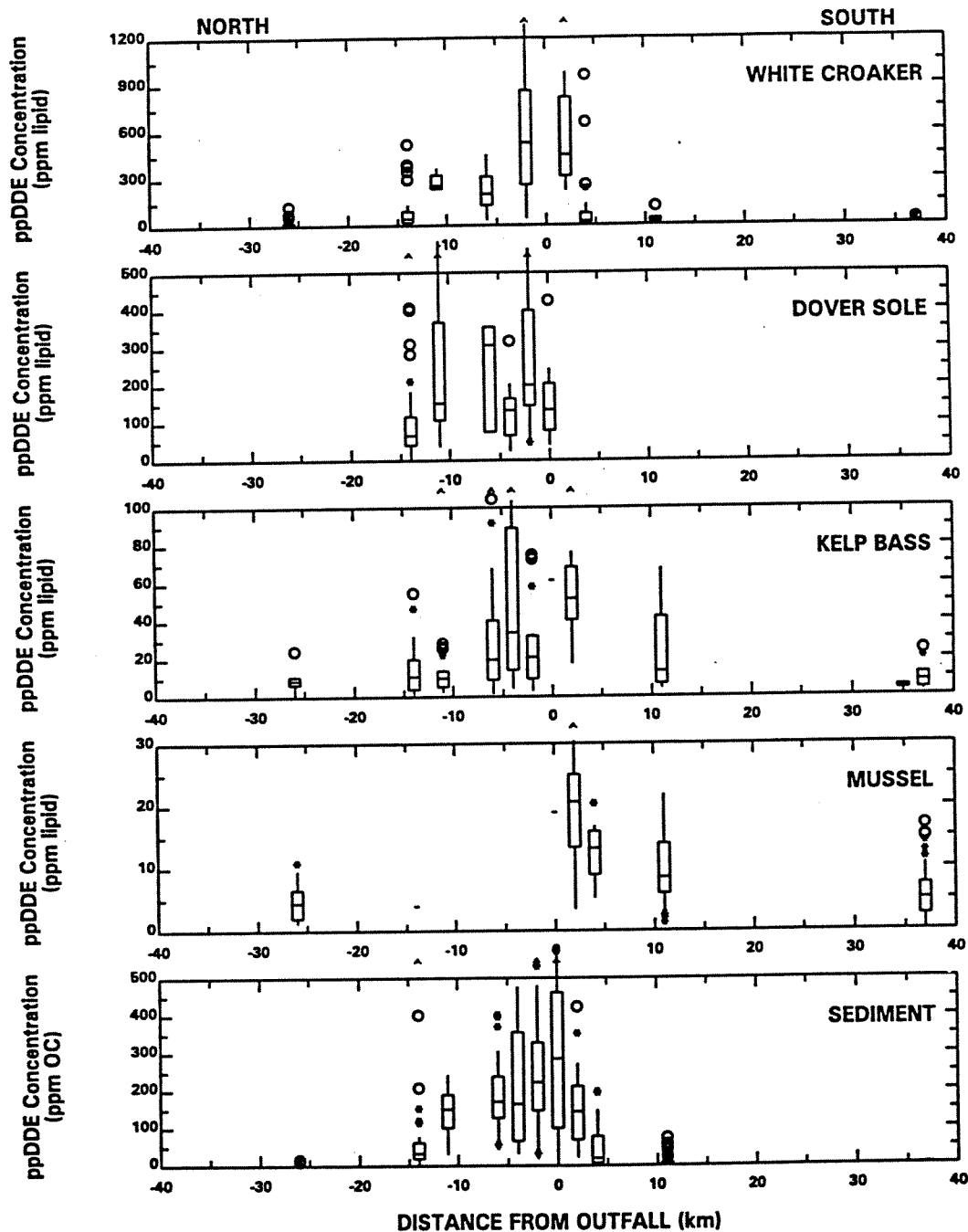
Figure B2-13. Observed p,p'DDE concentrations in fish, sediment and mussels collected from the Southern California Bight between 1985 and 1995. Concentrations in fish fillets and mussels are expressed as ppm lipid. Surficial sediment concentrations are expressed as ppm organic carbon. All data are plotted as a function of distance from the Los Angeles County Outfall (kilometer 0).
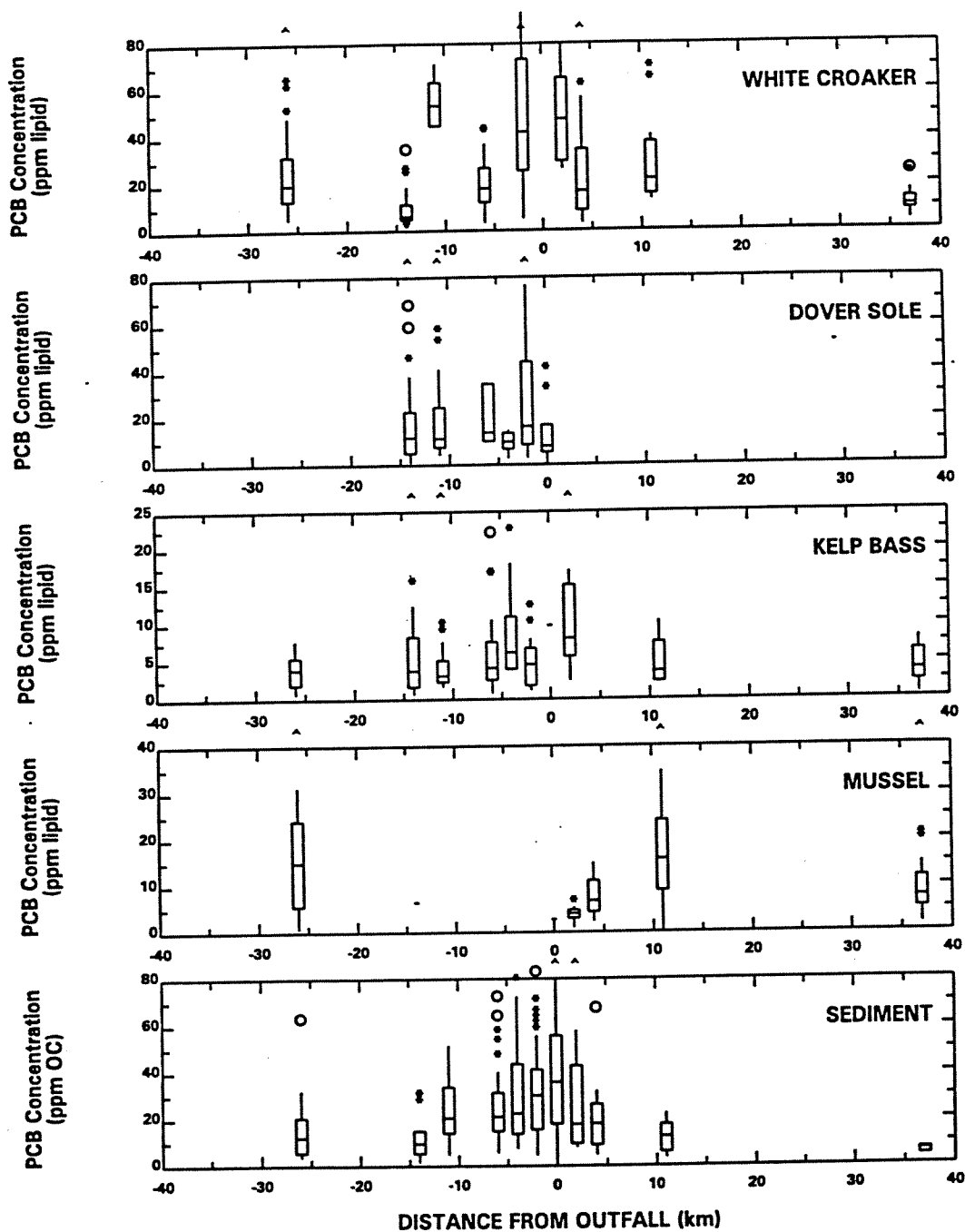
Figure B2-14. Observed total PCB concentrations in fish, sediment and mussels collected from the Southern California Bight between 1985 and 1995. Concentrations in fish fillets and mussels are expressed as ppm lipid. Surficial sediment concentrations are expressed as ppm organic carbon. All data are plotted as a function of distance from the Los Angeles County Outfall (kilometer 0).
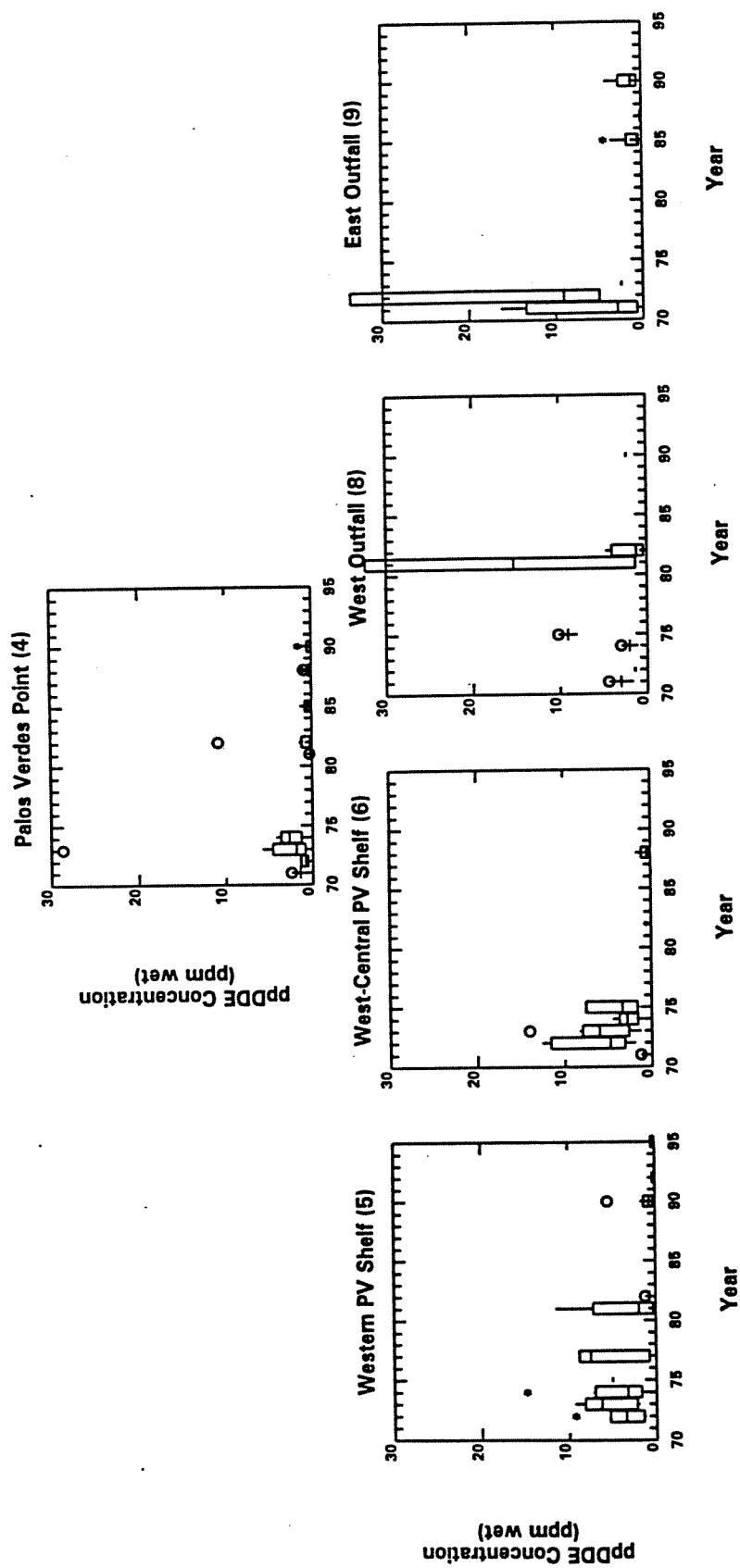
Figure B2-15. Temporal profiles of p,p'DDE concentrations in kelp bass for selected segments of the Southern California Bight (ppm wet weight).
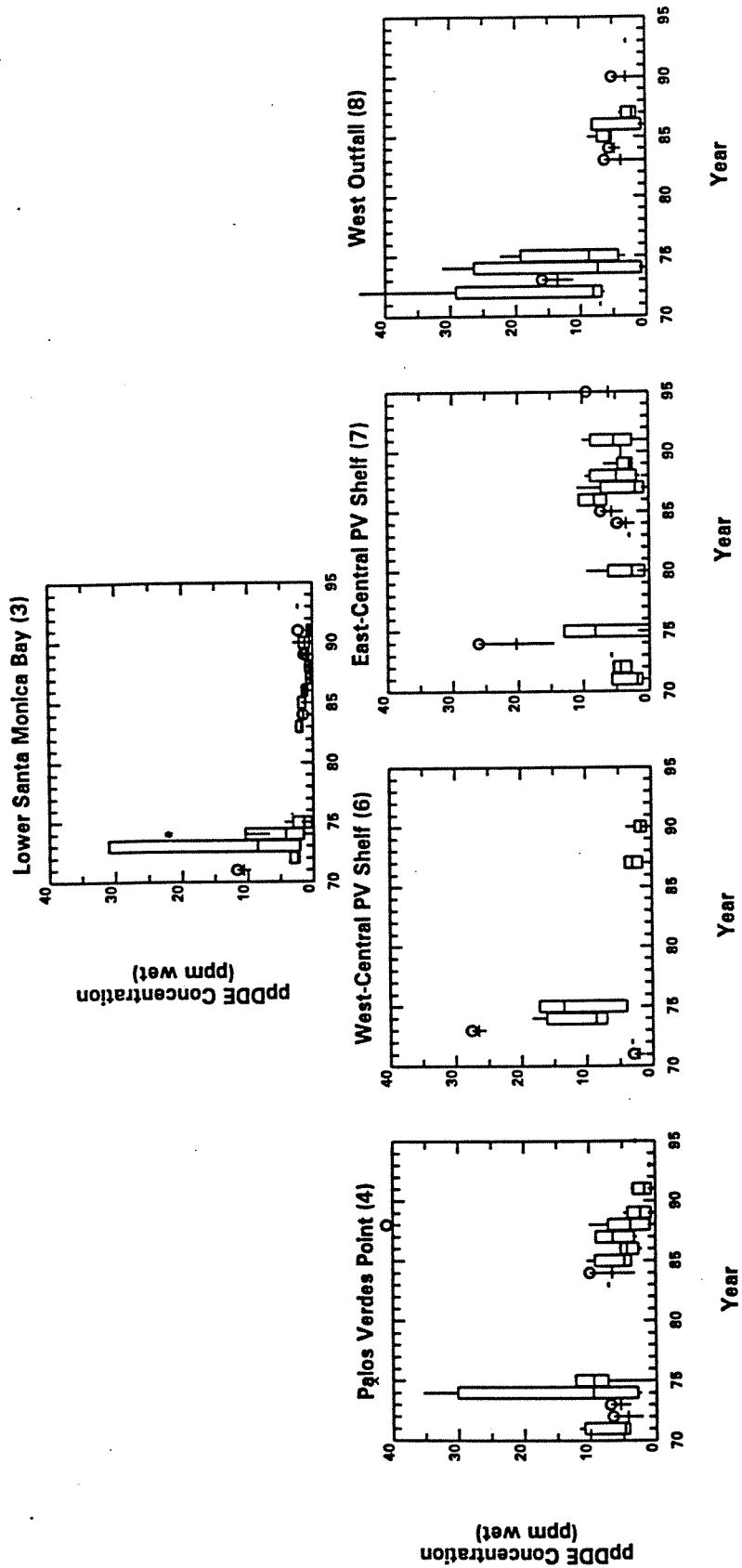
Figure B2-16. Temporal profiles of p,p'DDE concentrations in Dover sole for selected segments of the Southern California Bight (ppm wet weight).
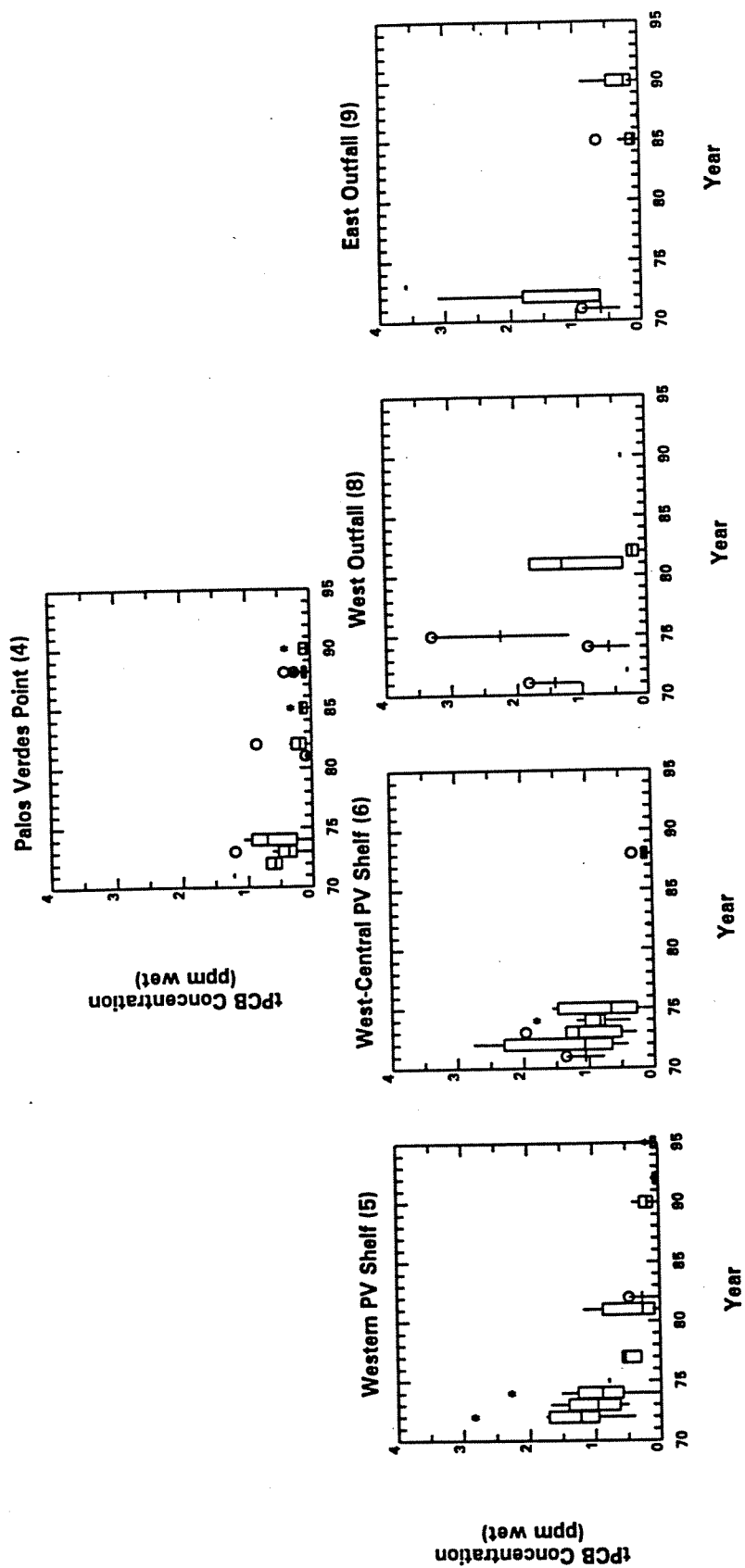
Figure B2-17. Temporal profiles of Total PCB concentrations in kelp bass for selected segments of the Southern California Bight (ppm wet weight).
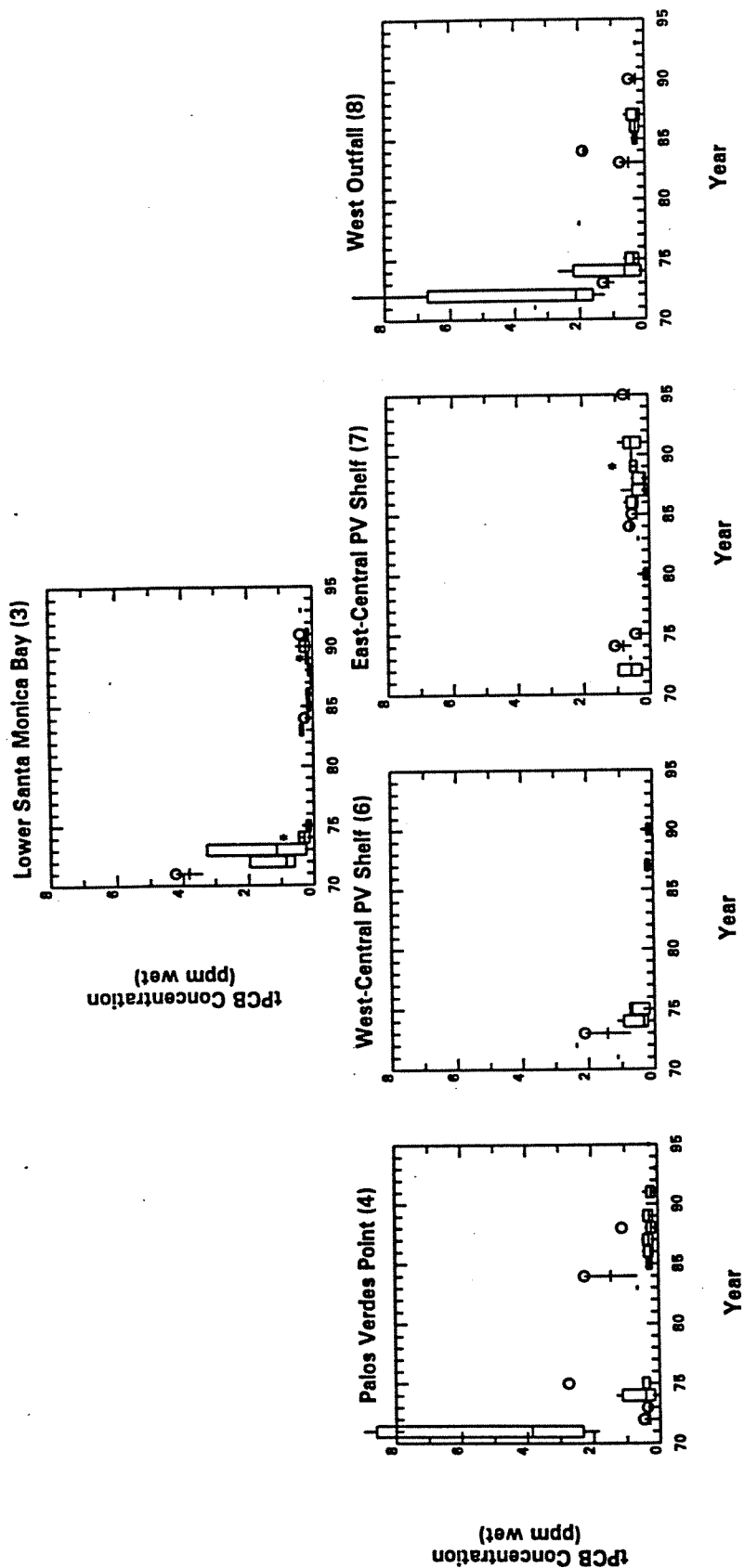
Figure B2-18. Temporal profiles of Total PCB concentrations in Dover sole for selected segments of the Southern California Bight (ppm wet weight).
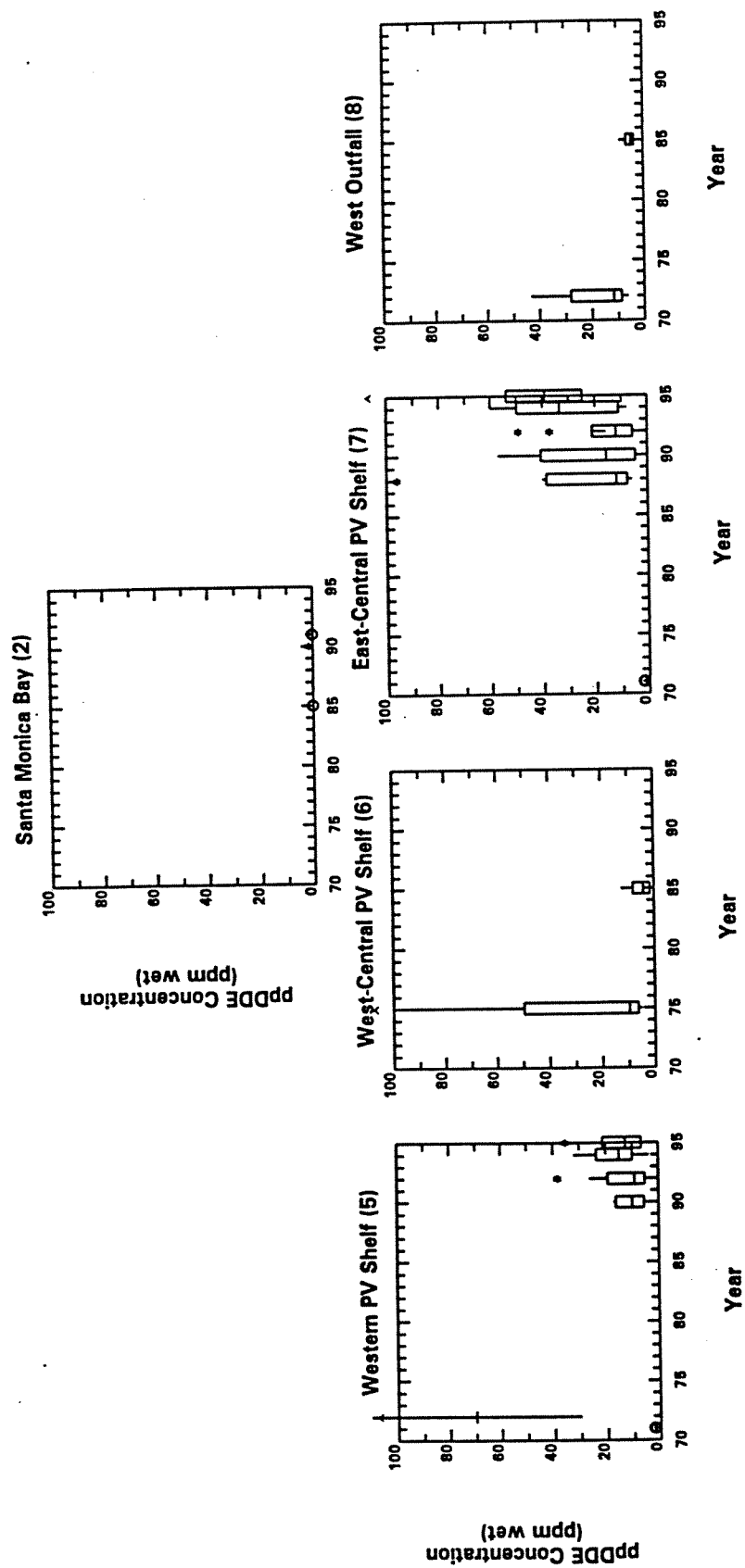
Figure B2-19. Temporal profiles of p,p'DDE concentrations in white croaker for selected segments of the Southern California Bight (ppm wet weight).
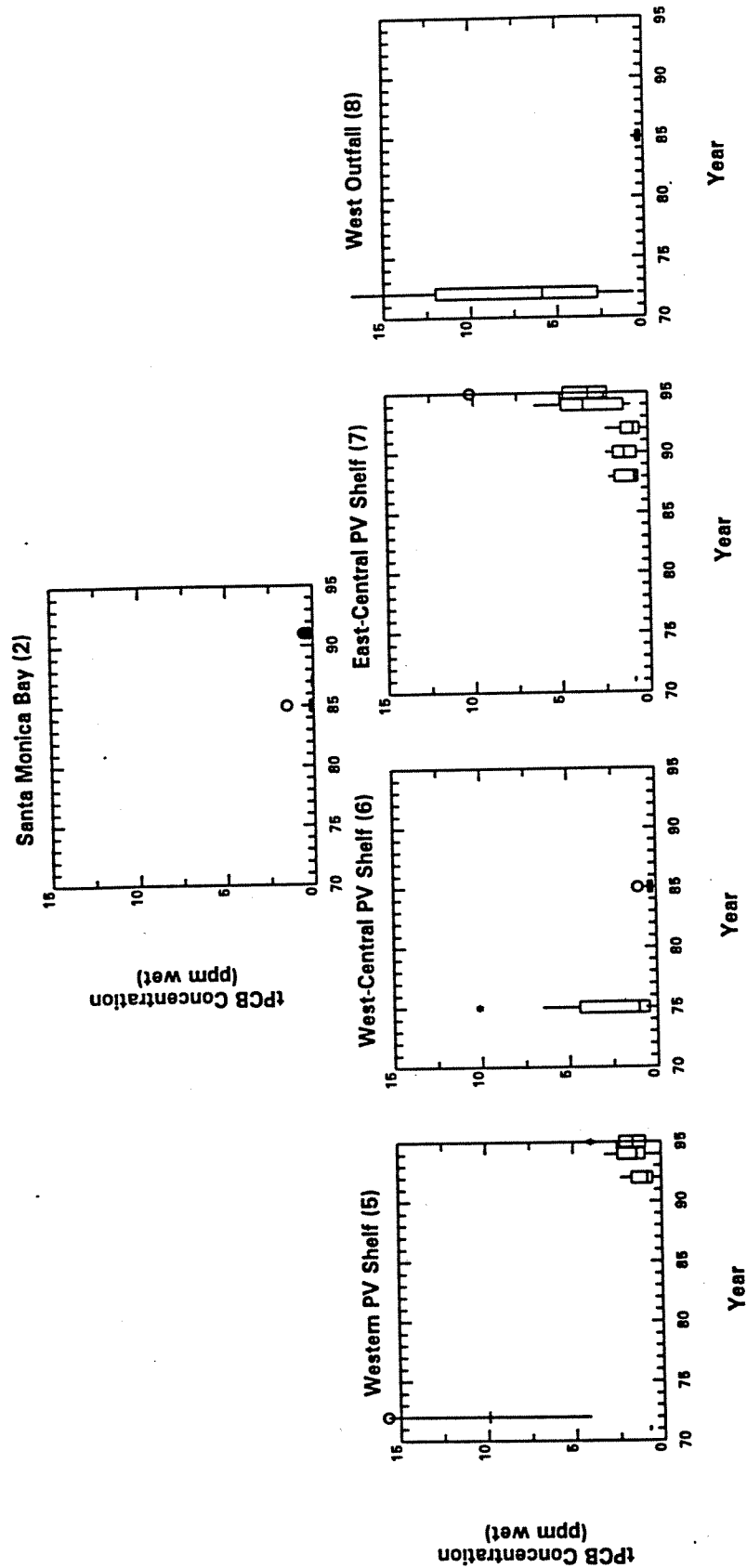
Figure B2-20. Temporal profiles of Total PCB concentrations in white croaker for selected segments of the Southern California Bight (ppm wet weight).

APPENDIX C

BOX PLOTS FOR FISH AND SEA LION
MODEL/DATA COMPARISONS
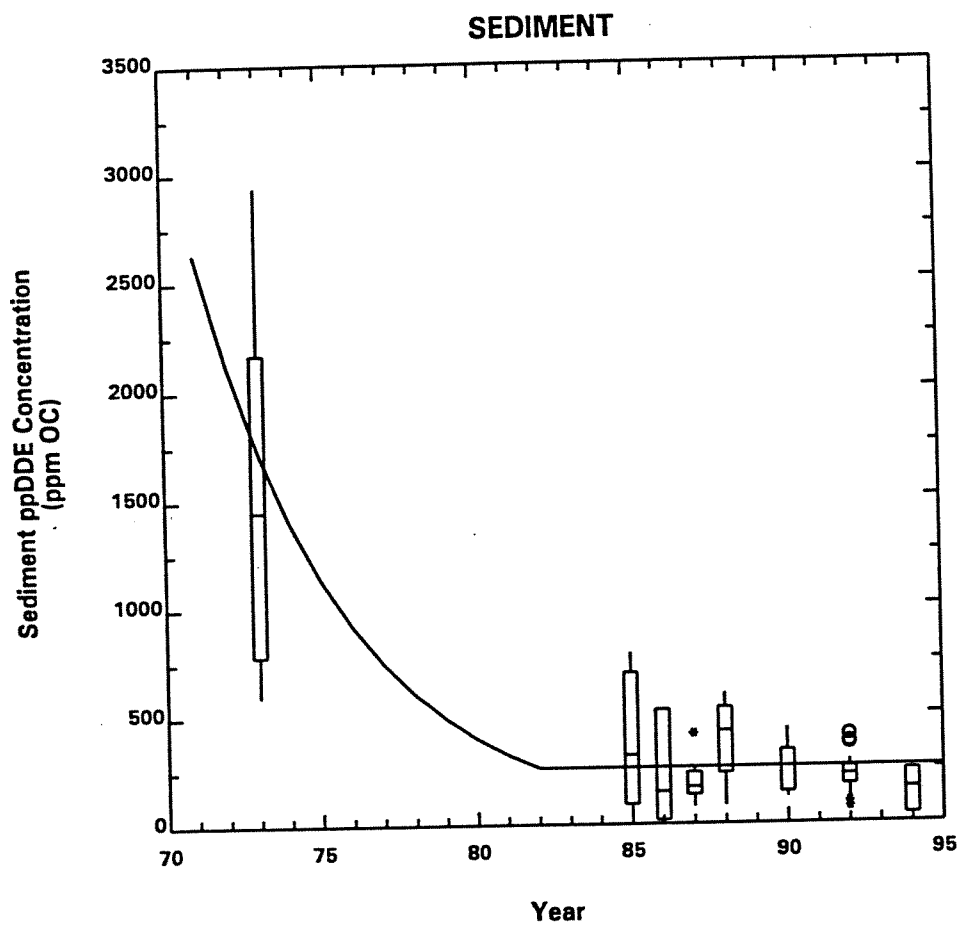
SEDIMENT

Figure C3-1. Temporal profile of p,p'DDE concentrations in surface sediments from zone of high contamination (segment 7; ppm organic carbon). Lines indicate exposure concentrations used in model.
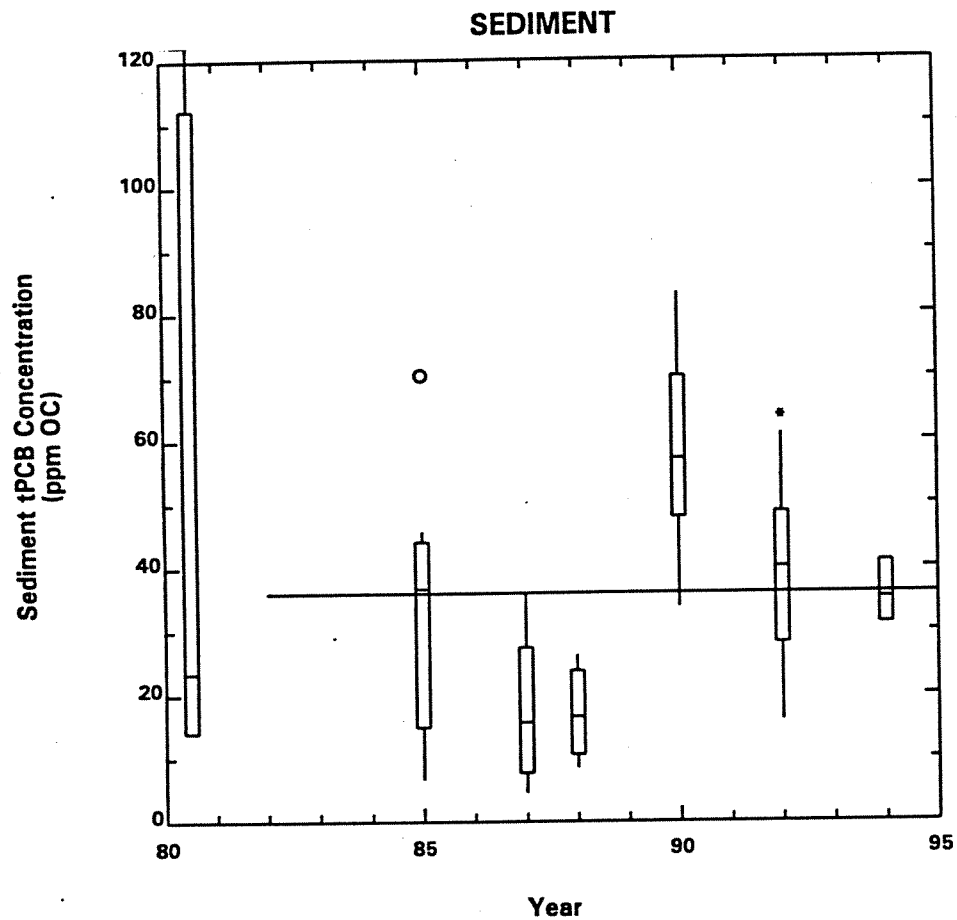
Figure C3-2. Temporal profile of Total PCB concentrations in surface sediments from zone of high contamination (segment 7; ppm organic carbon). Lines indicate exposure concentrations used in model.
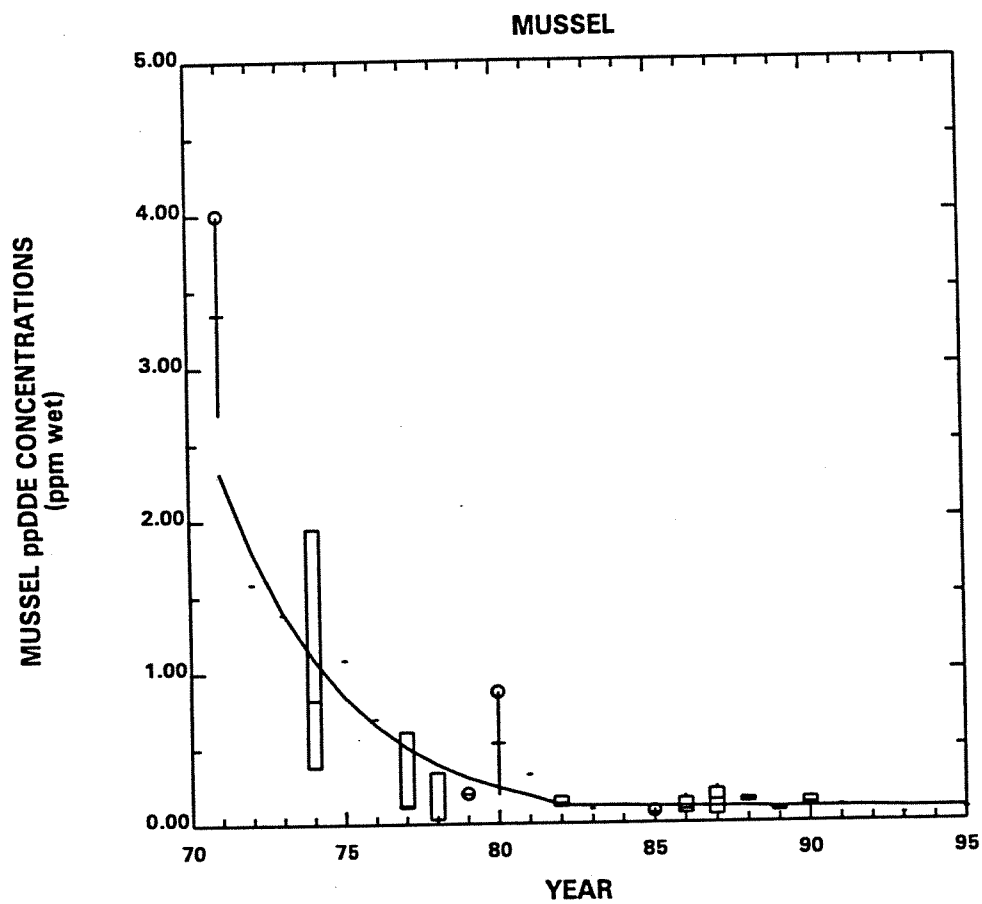
Figure C3-3. Temporal profile of p,p'DDE concentrations in mussels for the zone of high contamination (segments 8 and 9; ppm wet weight). Lines indicate exposure concentrations used in model.
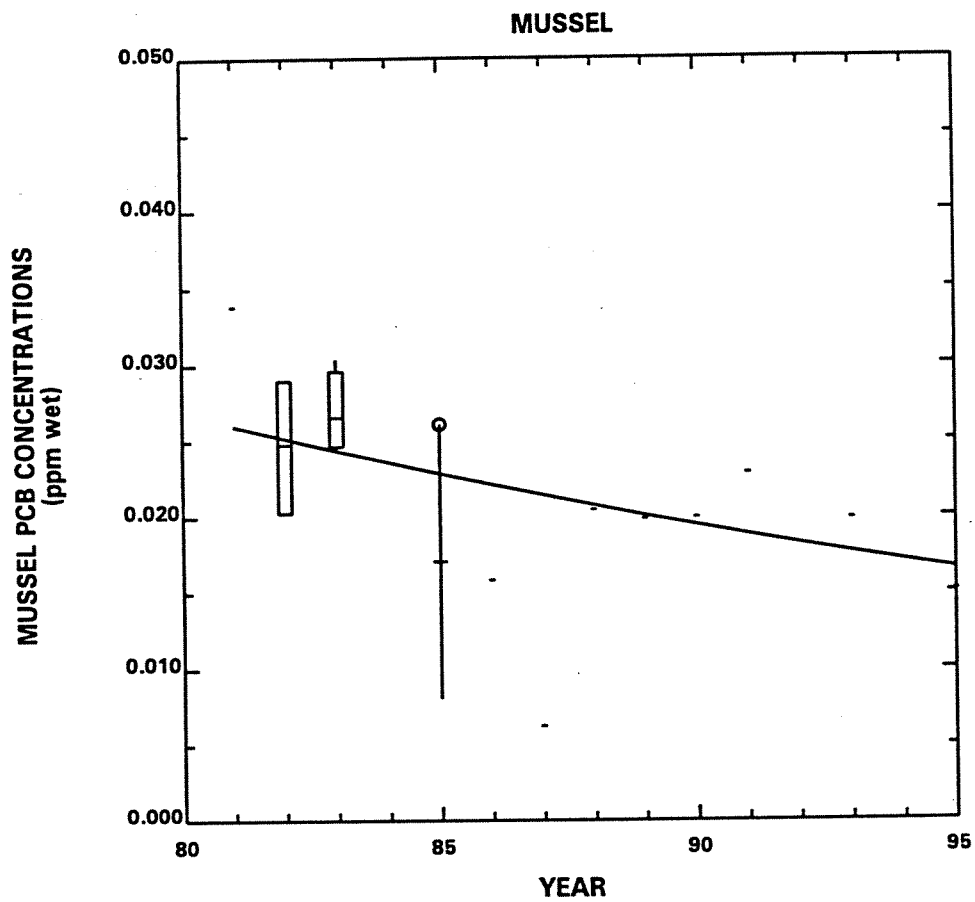
MUSSEL

Figure C3-4. Temporal profile of Total PCB concentrations in mussels for the zone of high contamination (segments 8 and 9; ppm wet weight). Lines indicate exposure concentrations used in model.

GROWTH INFORMATION

WHITE CROAKER IN THE SCB

Length/Age Relationship

Length Distribution

Weight/Length Relationship

Percent Lipid/Length Relationship

Weight/Age Relationship

Percent Dry Weight/Length Relationship
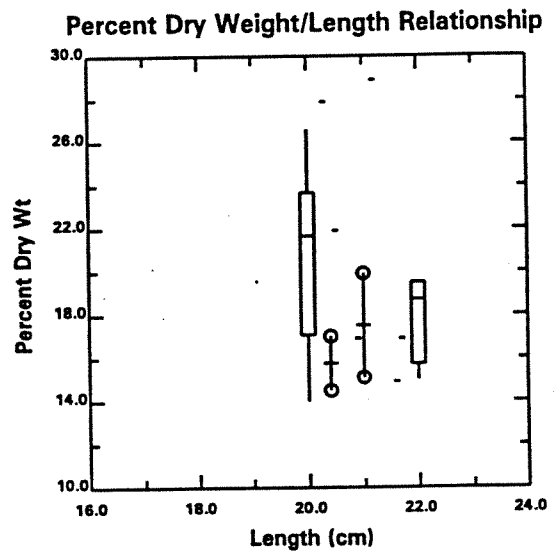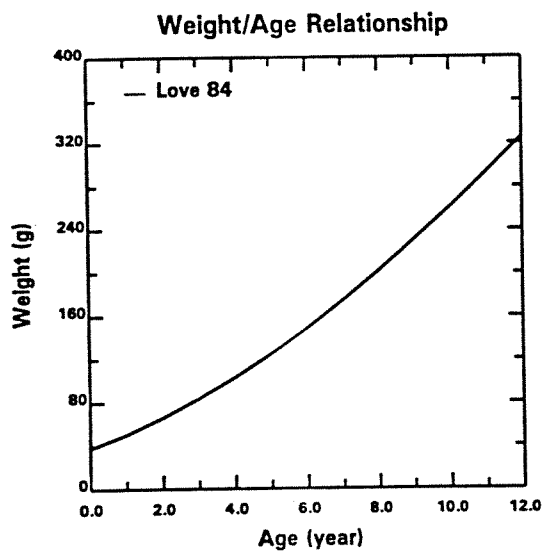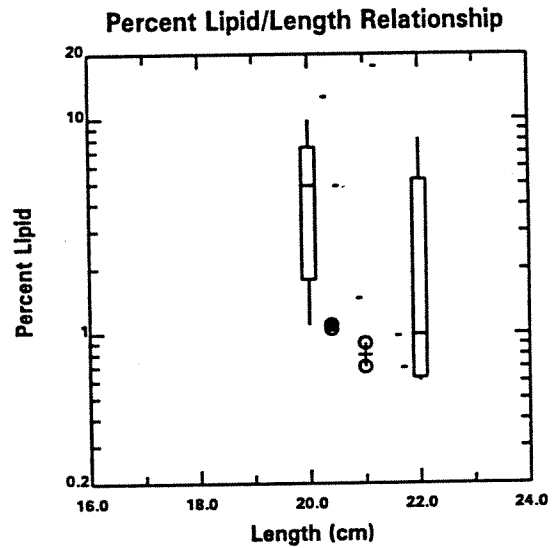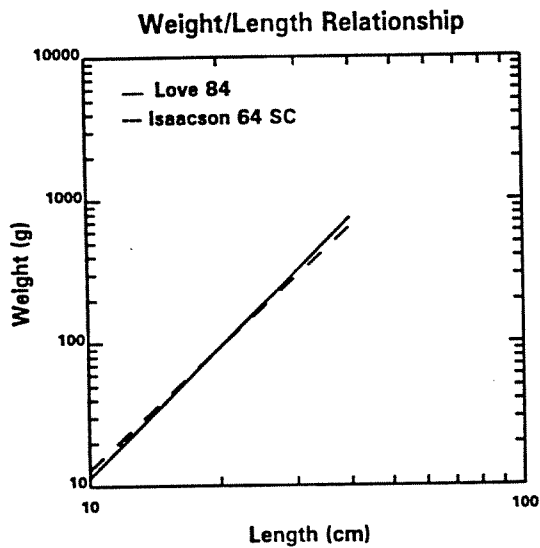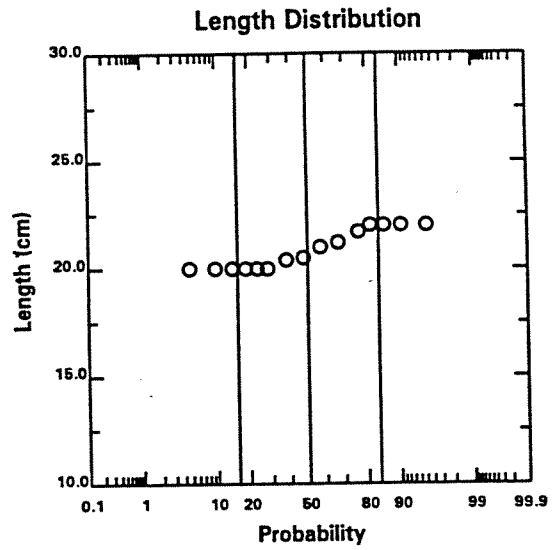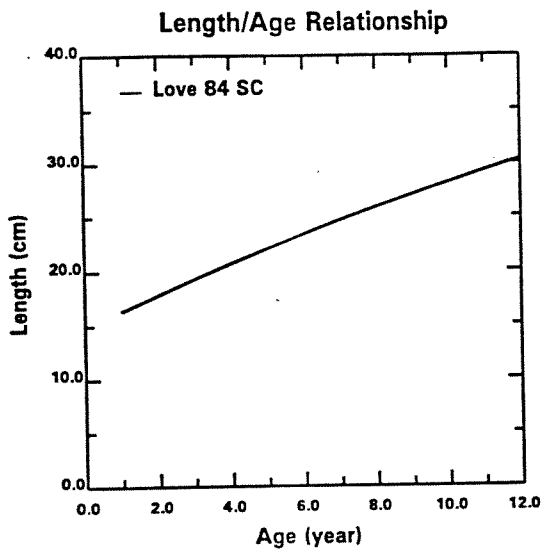
Figure C3-9. White croaker growth and body composition data.

WHITE CROAKER

WHITE CROAKER

·SOUTHERN CALIF BIGHT

FOOD CHAIN MODEL

SEGMENT 7

ZOOPLANKTON

CAPITELLIDS

PHYTOPLANKTON (ppm OC)

SEDIMENT (ppm OC)

Year

pp-DDE (ppm lipid)

WATER COLUMN (ppm Water)

Year

Figure C3-11. Computed lipid normalized p,p'DDE concentrations (lines) and data (boxes) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
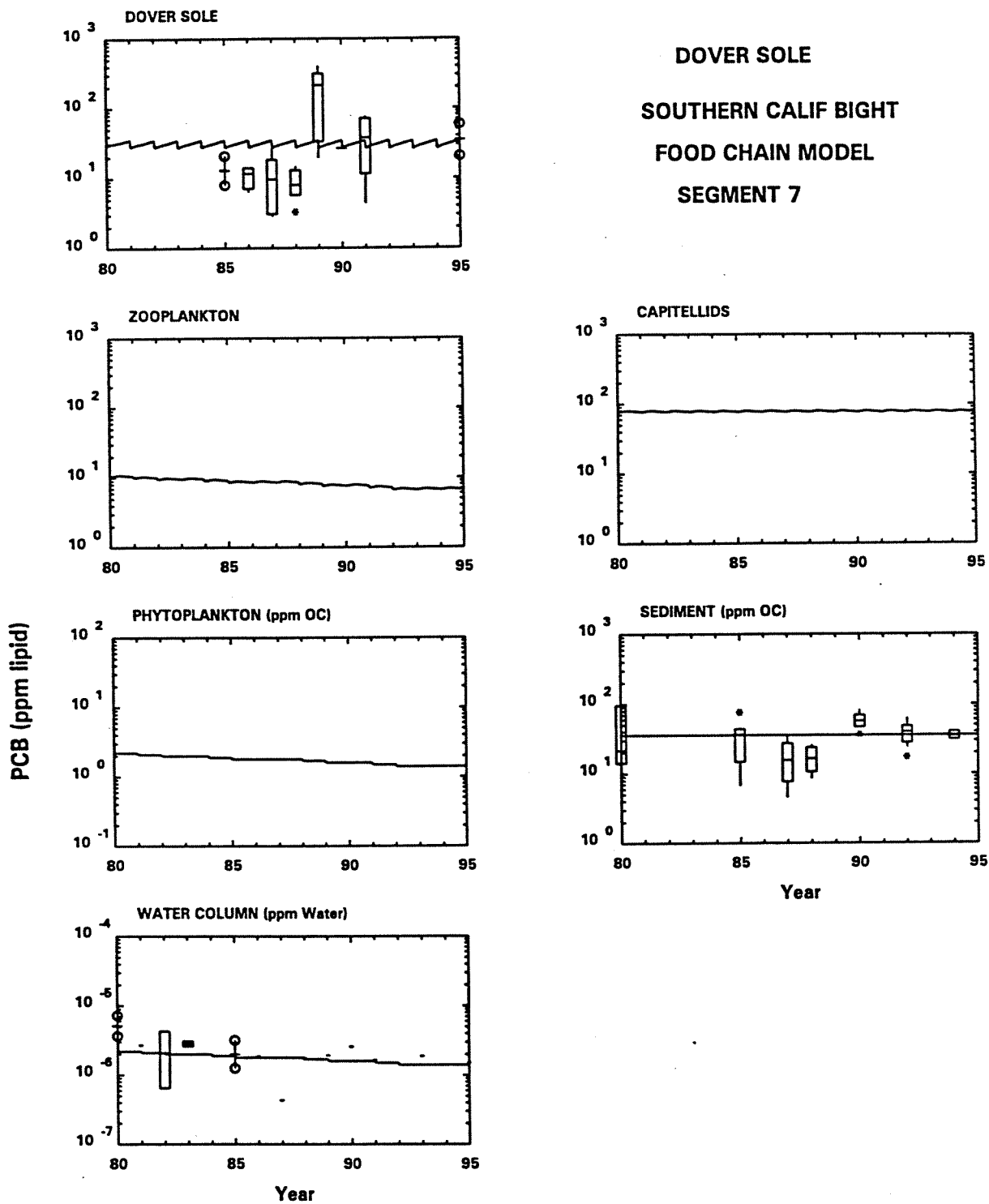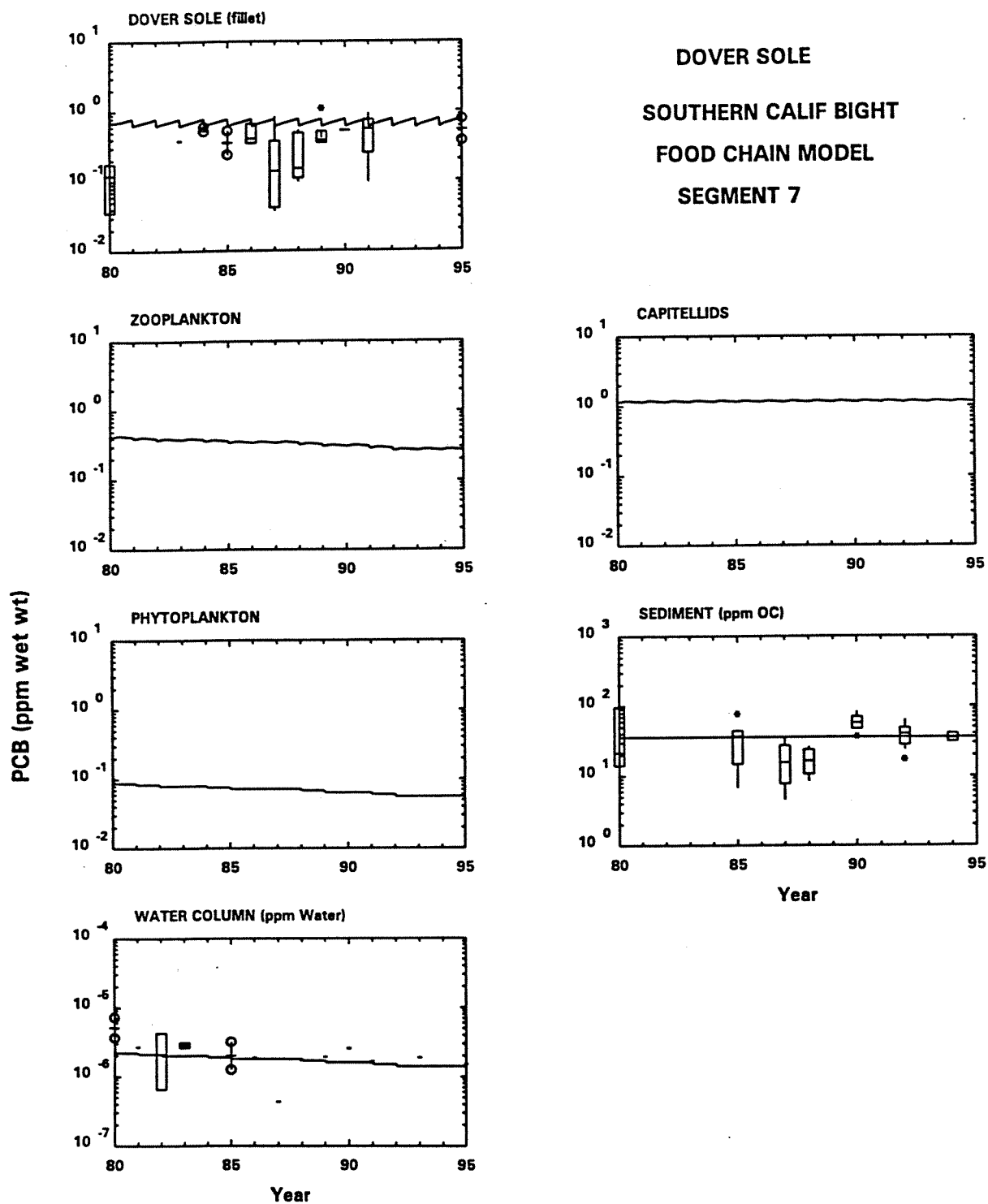
Figure C3-12. Computed wet weight p,p'DDE concentrations (lines) and data (boxes) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
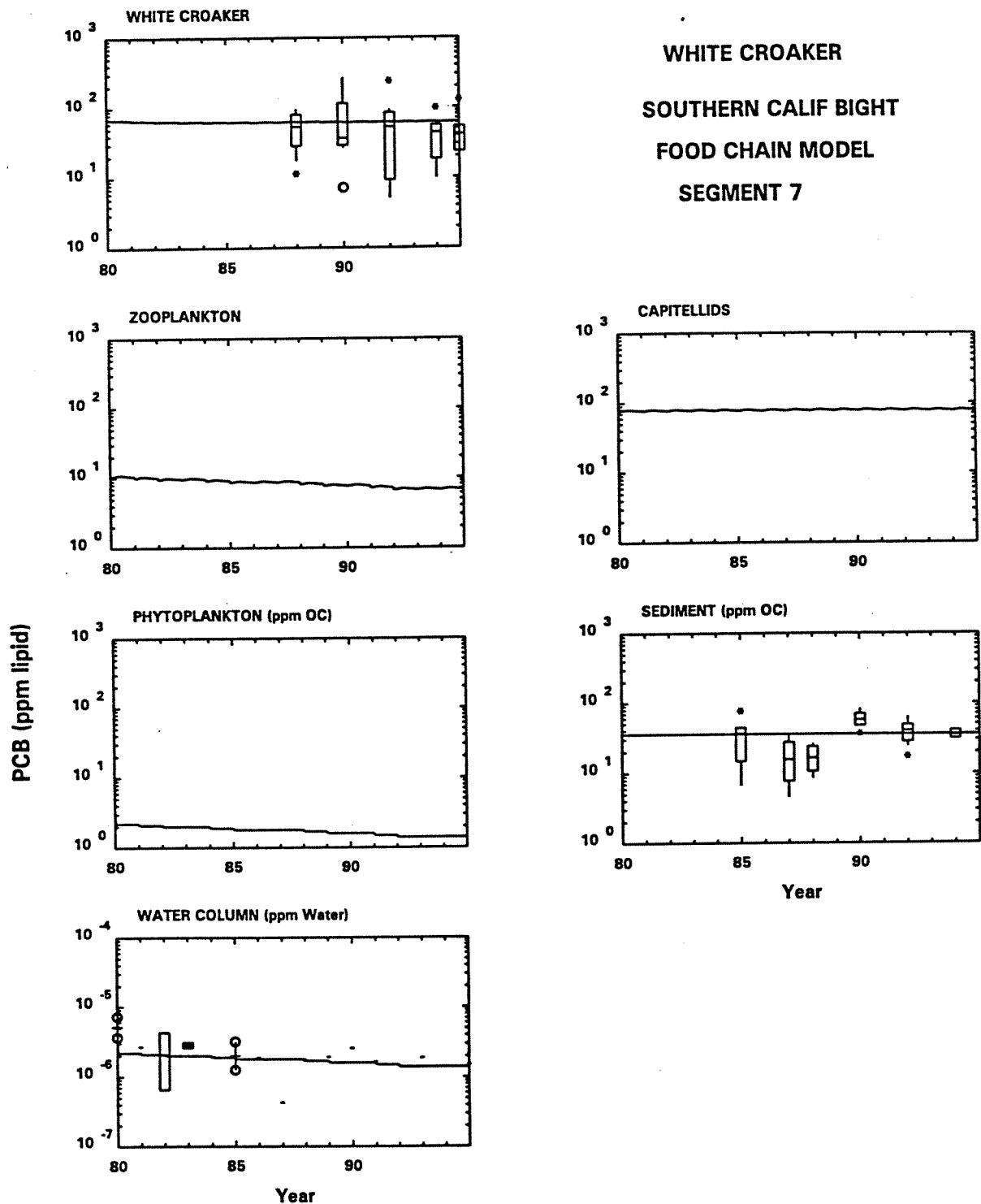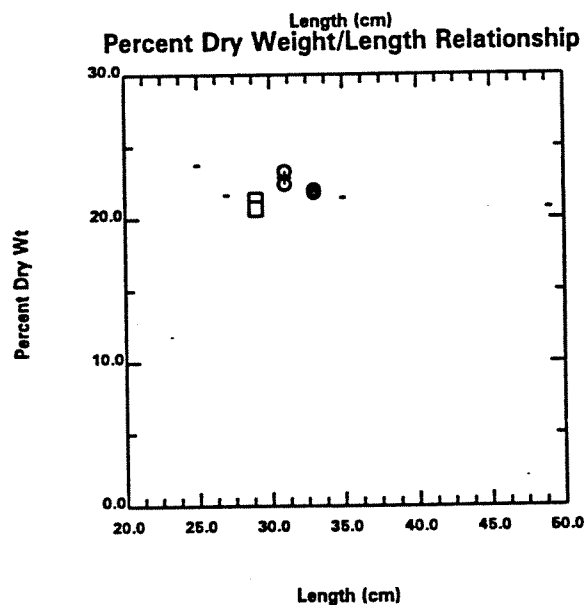
Figure C3-13. Computed lipid normalized Total PCB concentrations (lines) and data (boxes) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
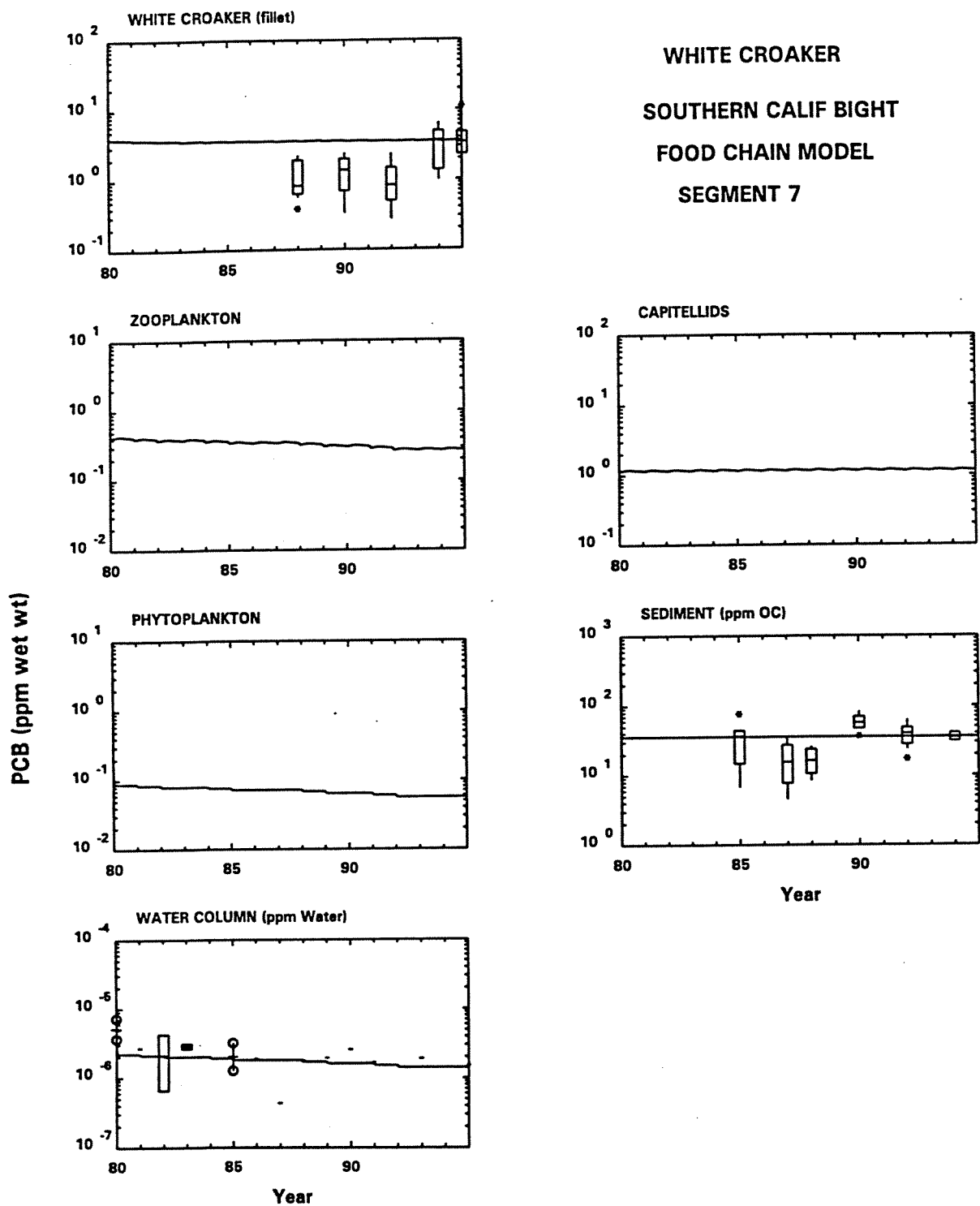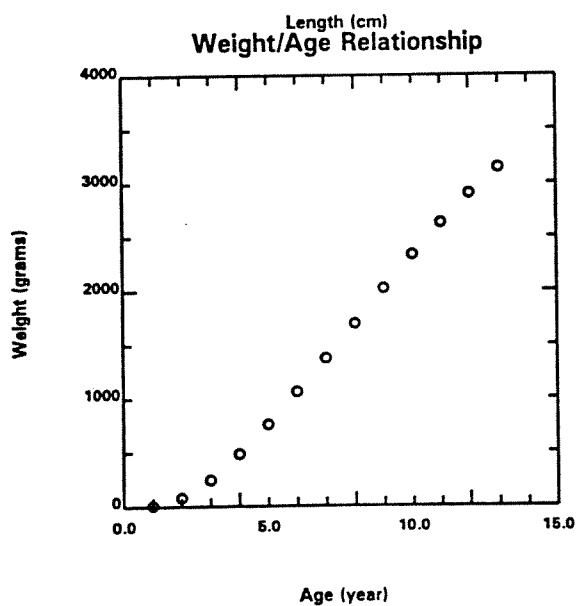
Figure C3-14. Computed wet weight Total PCB concentrations (lines) and data (boxes) for the white croaker food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
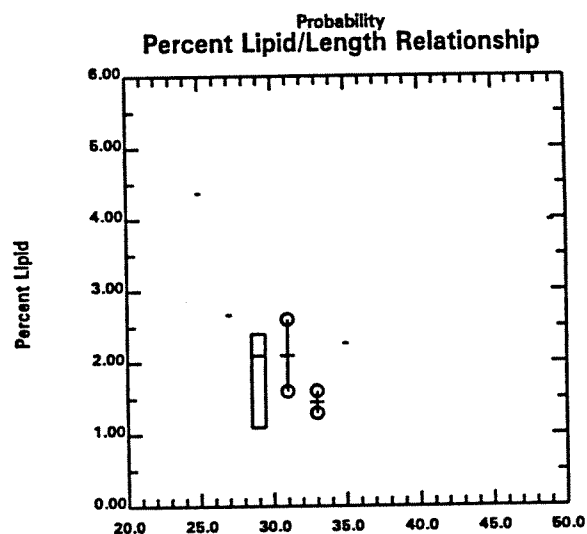
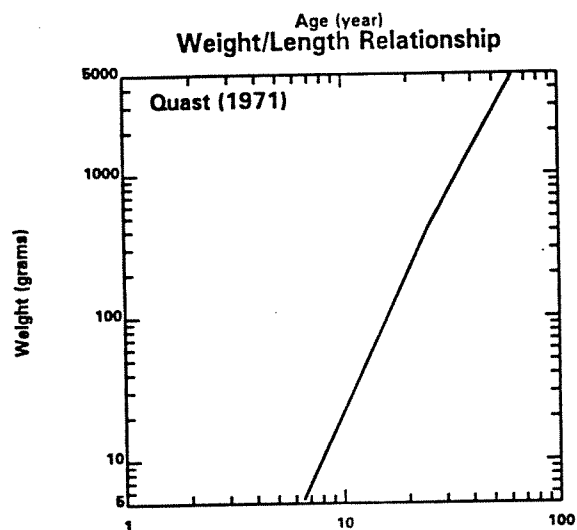# GROWTH INFORMATION

DOVER SOLE IN THE SCB

Length/Age Relationship

Length Distribution

Weight/Length Relationship

Percent Lipid/Length Relationship

Weight/Age Relationship

Percent Dry Weight/Length Relationship
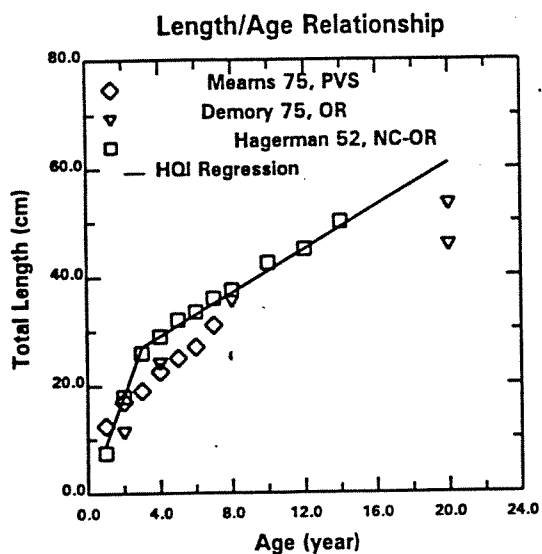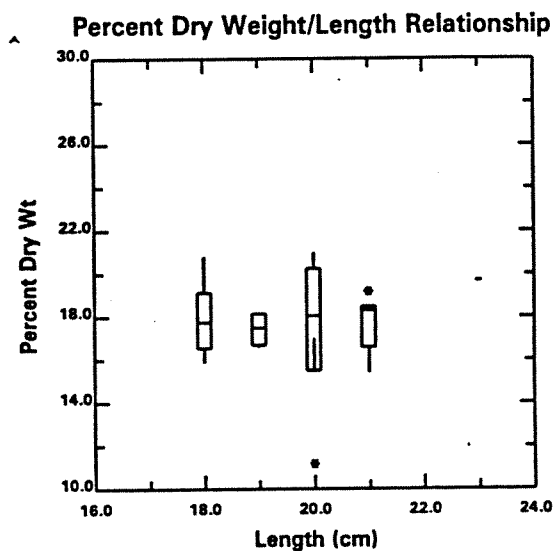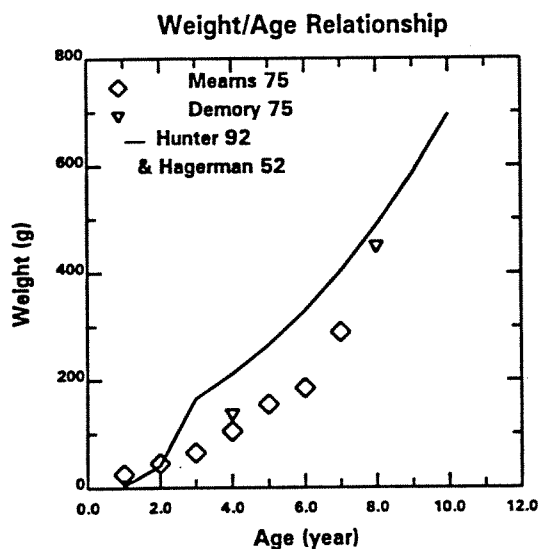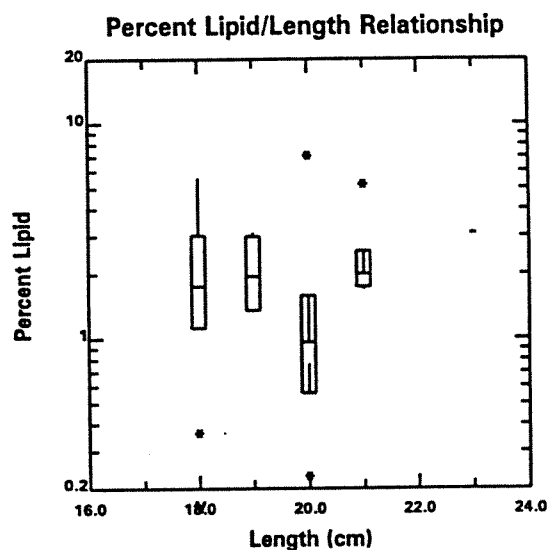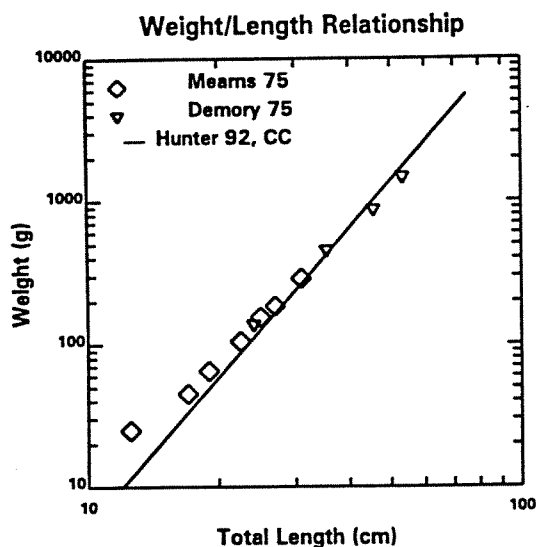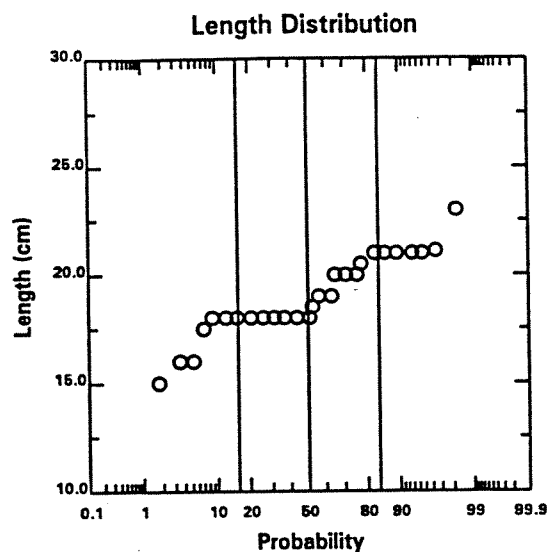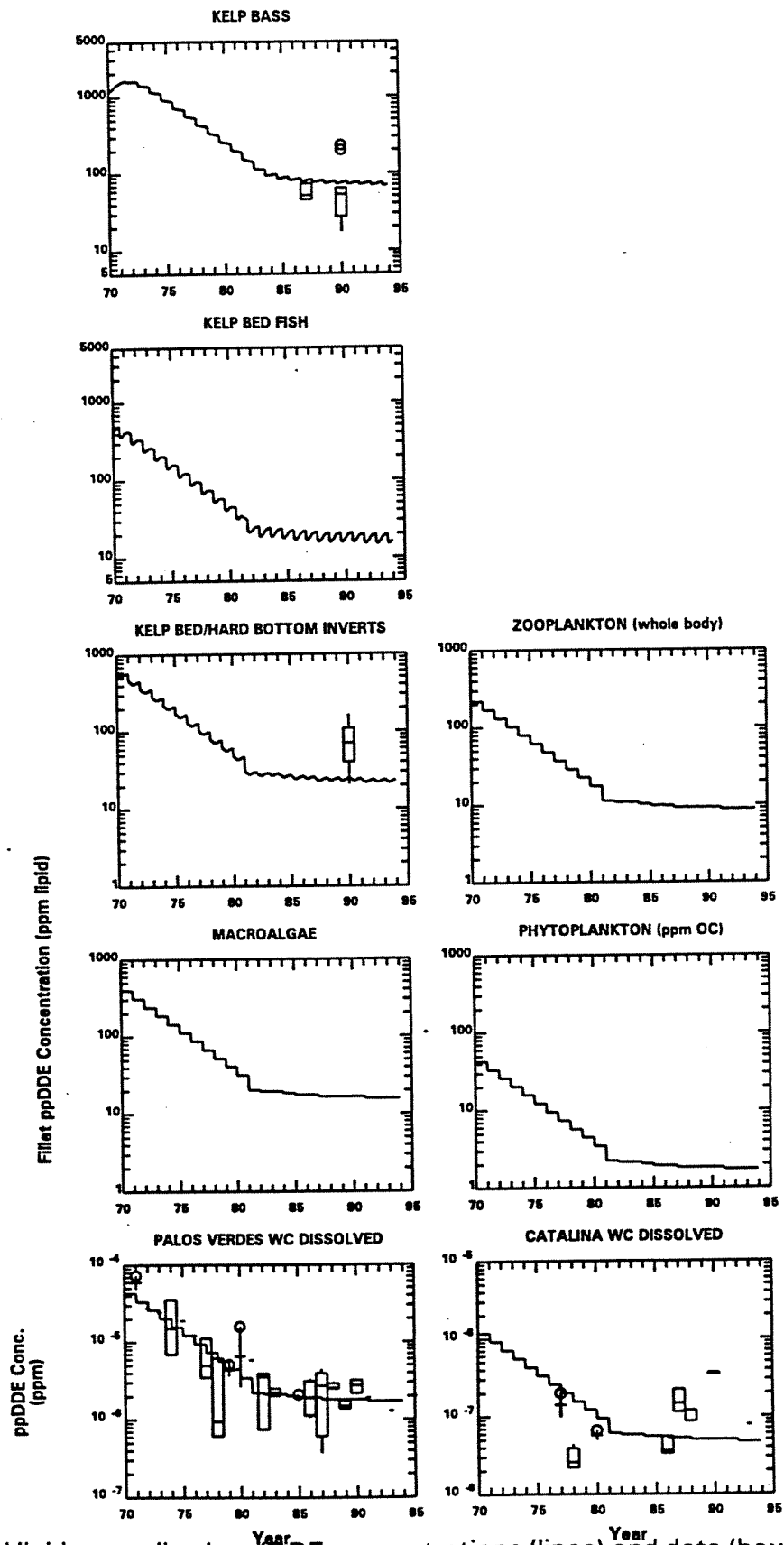

Figure C3-16. Dover sole growth and body composition data.
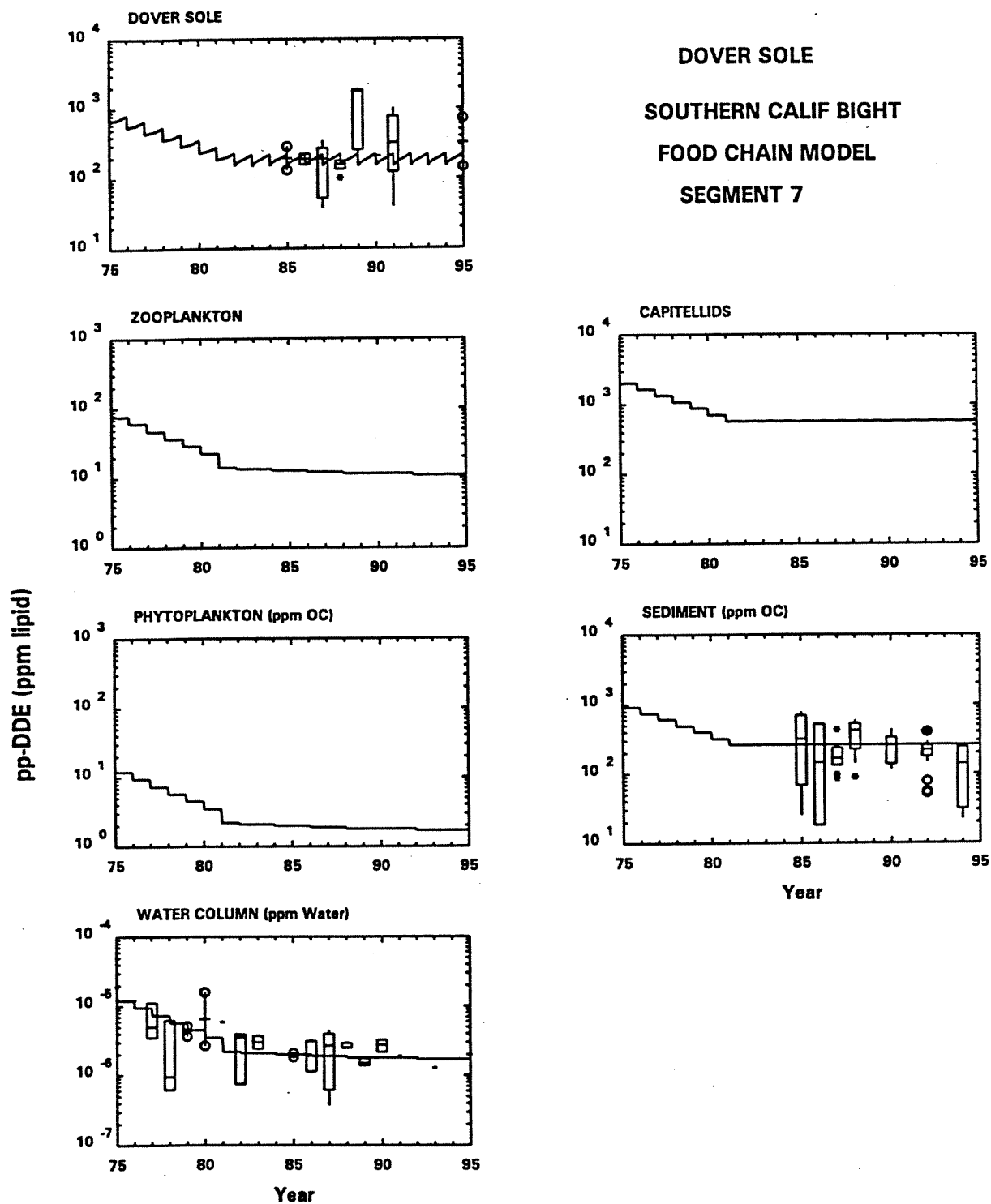
Figure C3-18. Computed lipid normalized p,p'DDE concentrations (lines) and data (boxes) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
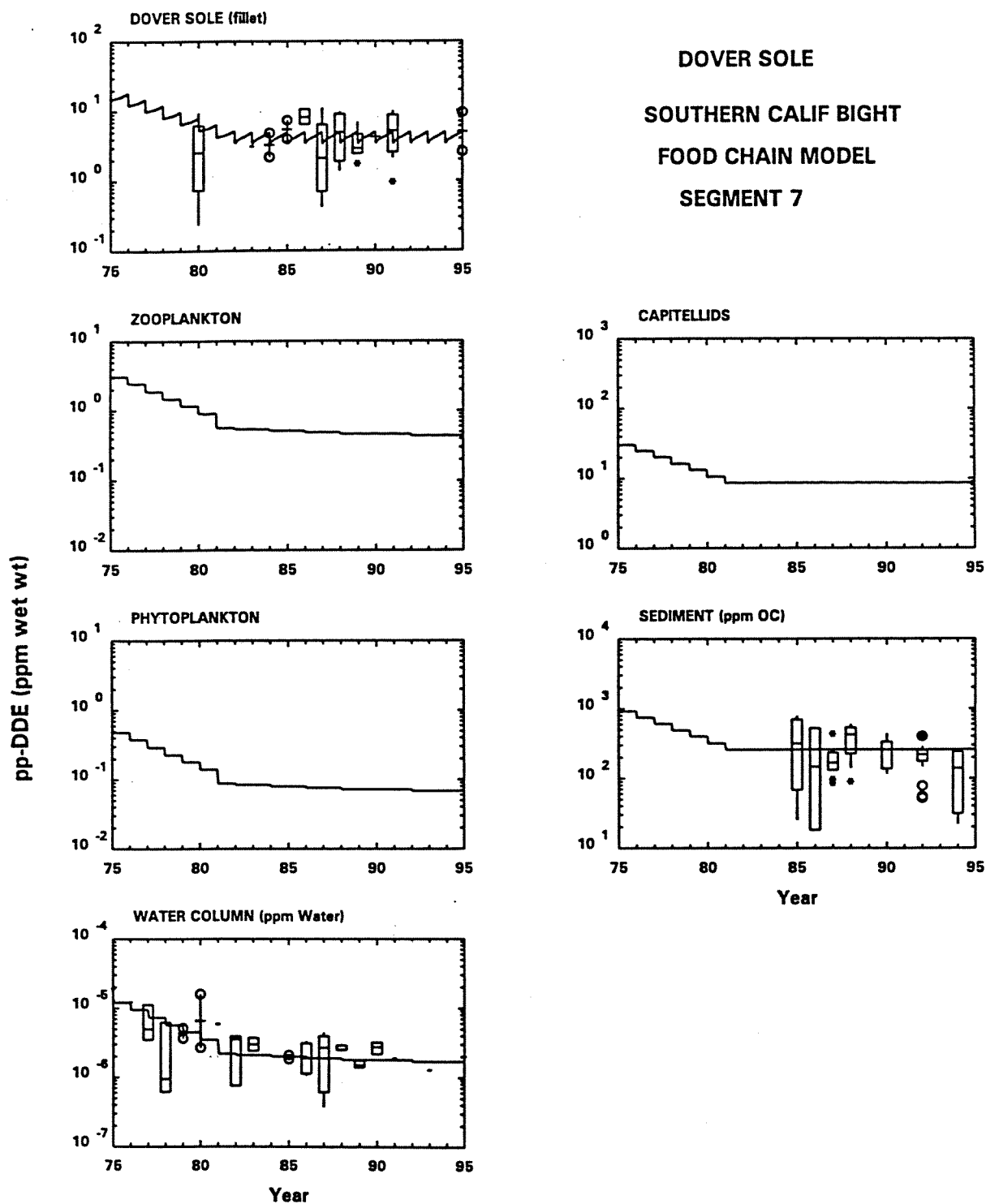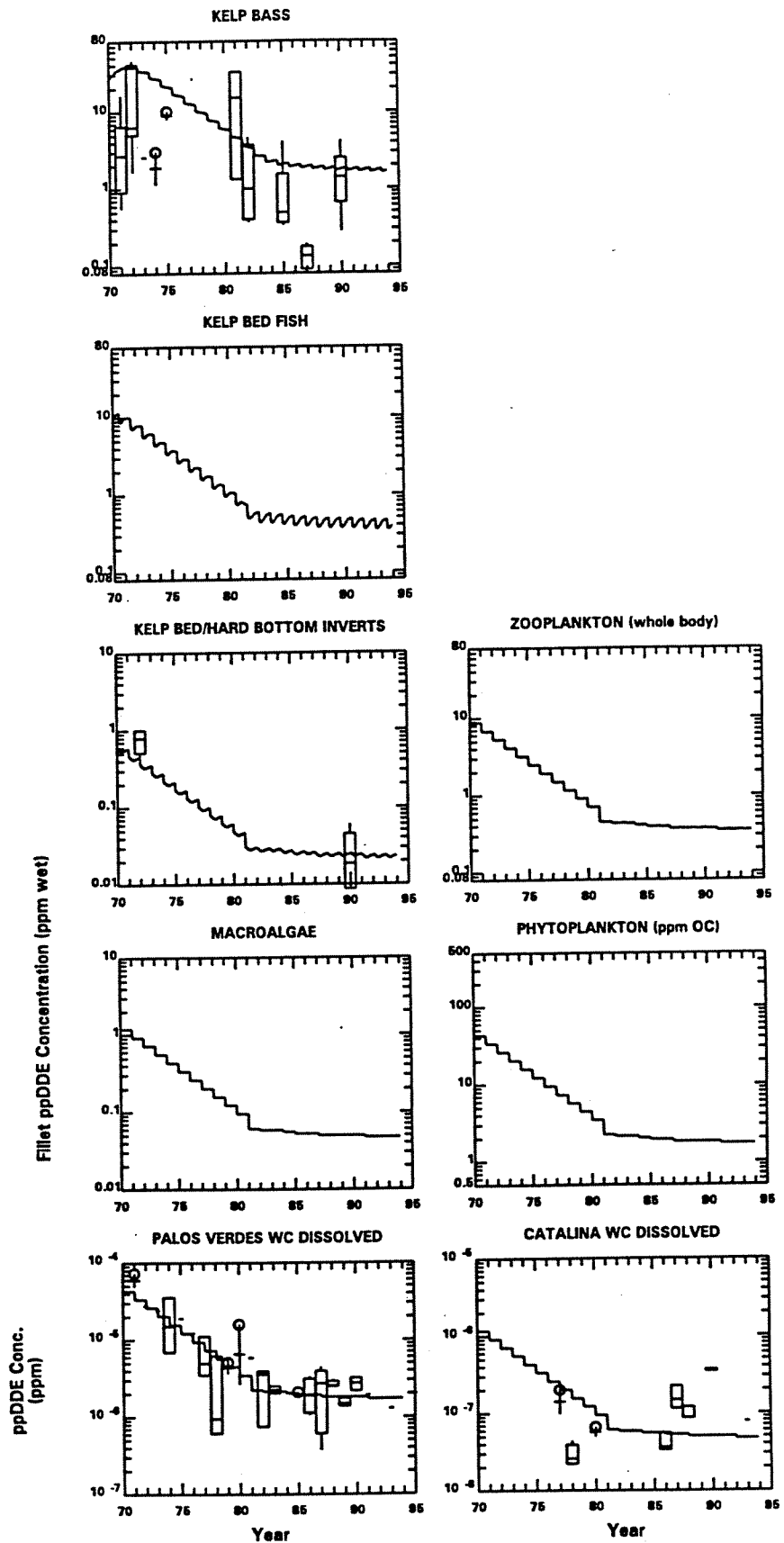
Figure C3-19. Computed wet weight p,p'DDE concentrations (lines) and data (boxes) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).
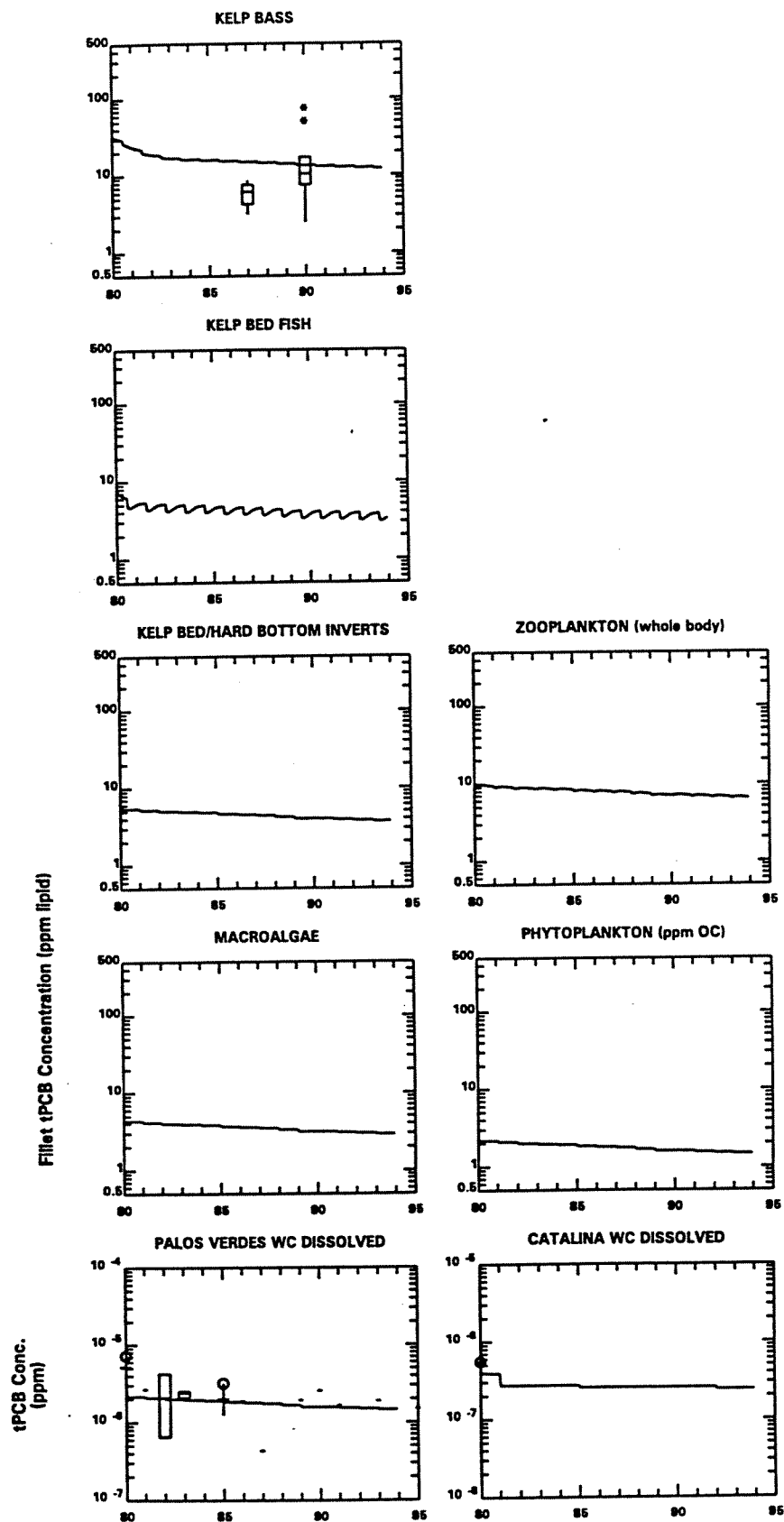
Figure C3-20. Computed lipid normalized Total PCB concentrations (lines) and data (boxes) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).

Figure C3-21. Computed wet weight Total PCB concentrations (lines) and data (boxes) for the Dover sole food web and water column and sediment exposure concentrations in East-Central Palos Verdes Shelf (Segment 7).

GROWTH INFORMATION

KELP BASS IN THE SCB

Length/Age Relationship

Length Distribution

Weight/Length Relationship

Percent Lipid/Length Relationship

Weight/Age Relationship

Percent Dry Weight/Length Relationship
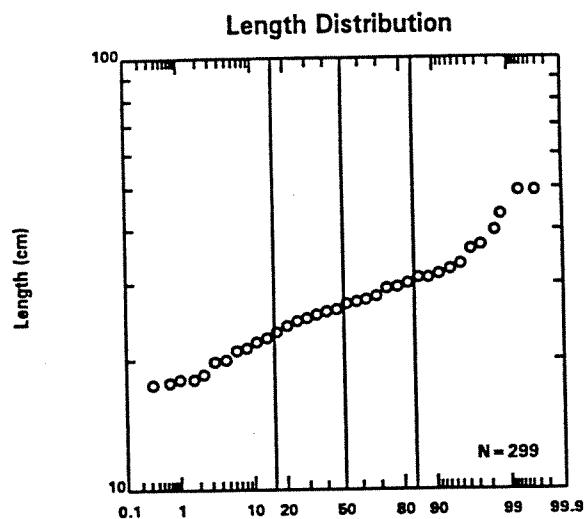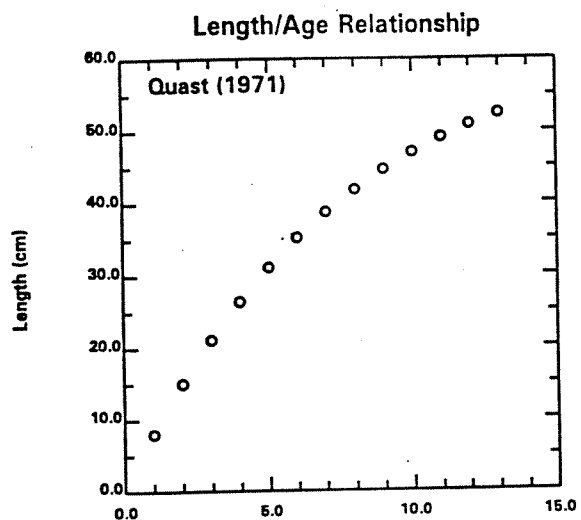
Figure C3-24. Kelp bass growth and body composition data.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

p,p - DDE

Figure C3-26. Computed lipid normalized p,p DDE concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

p,p - DDE
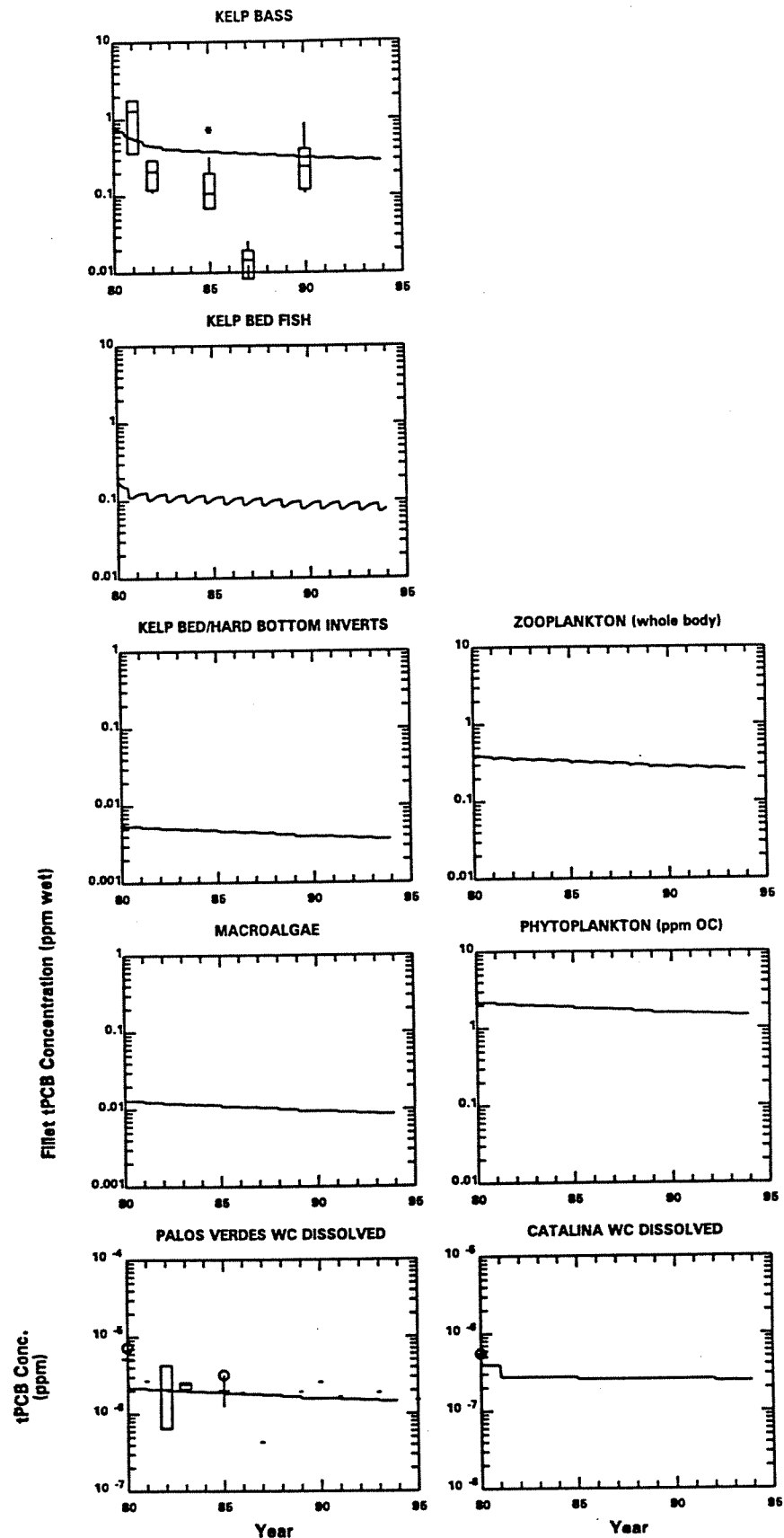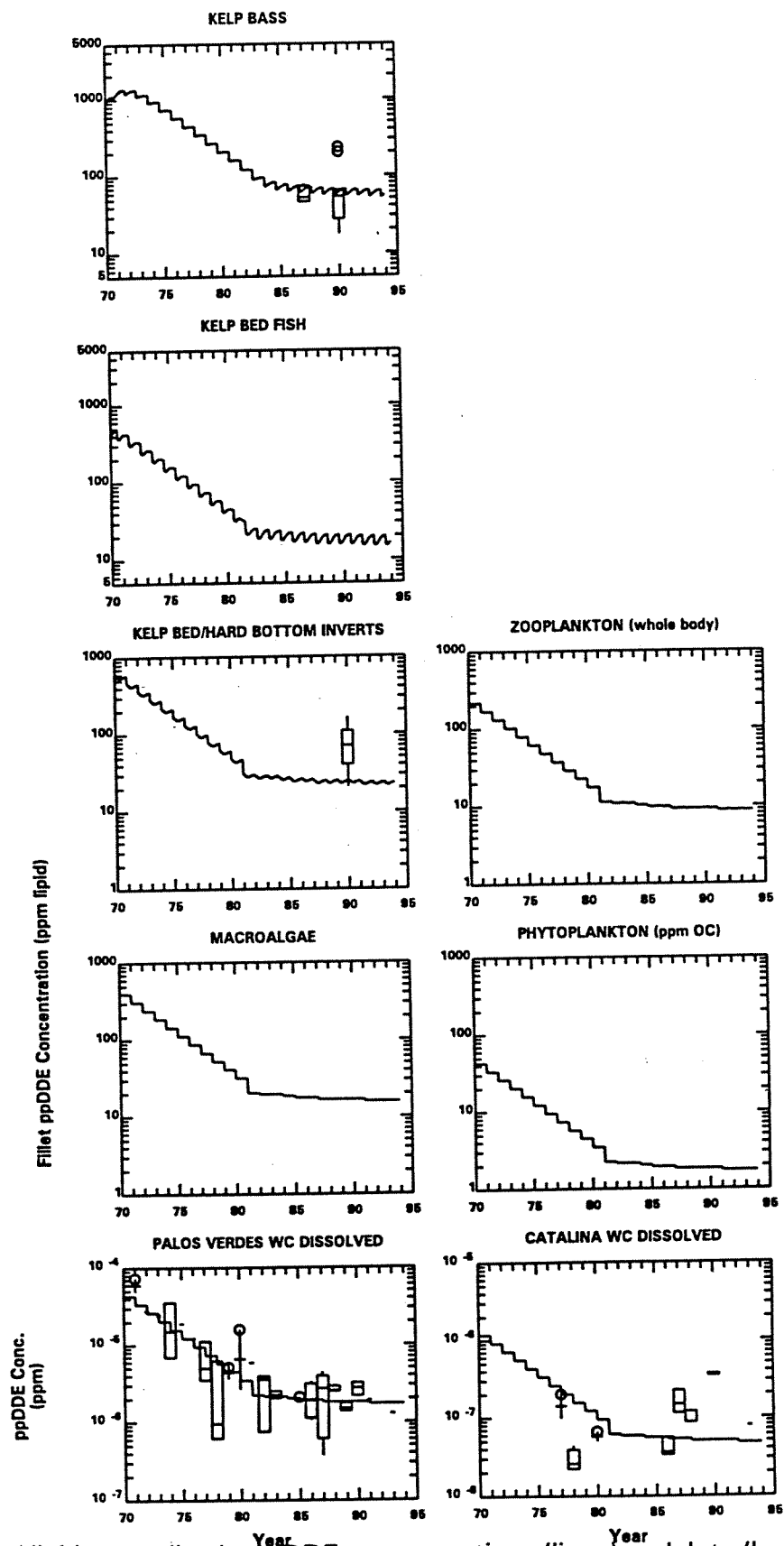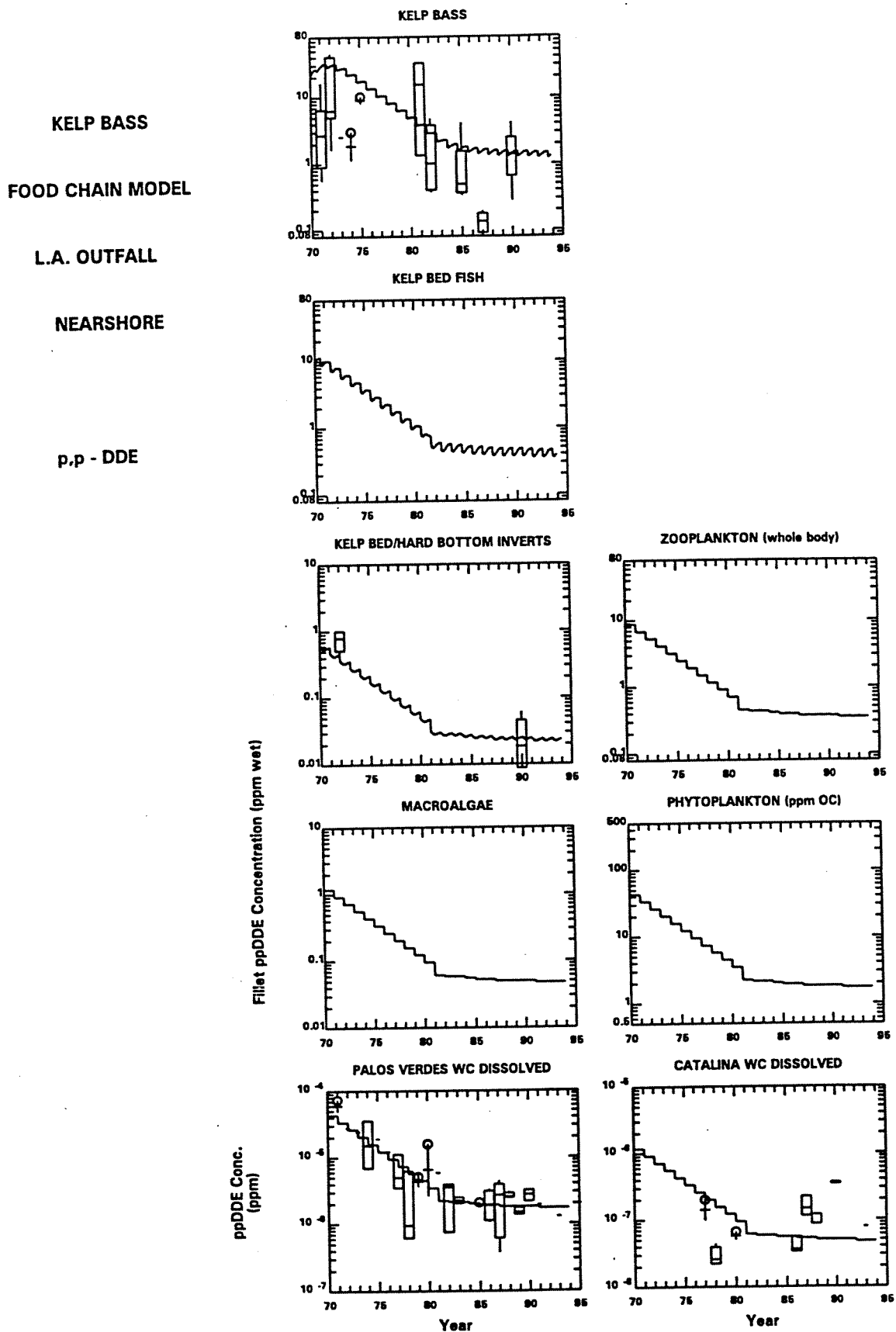

Figure C3-27. Computed wet weight p,p'DDE concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

TOTAL PCBs

Figure C3-28. Computed lipid normalized Total PCB concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario.

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

RUN 60

TOTAL PCBs

Figure C3-29. Computed wet weight Total PCB concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). No migration scenario.

Figure C3-30. Computed lipid normalized p,p'DDE concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island).
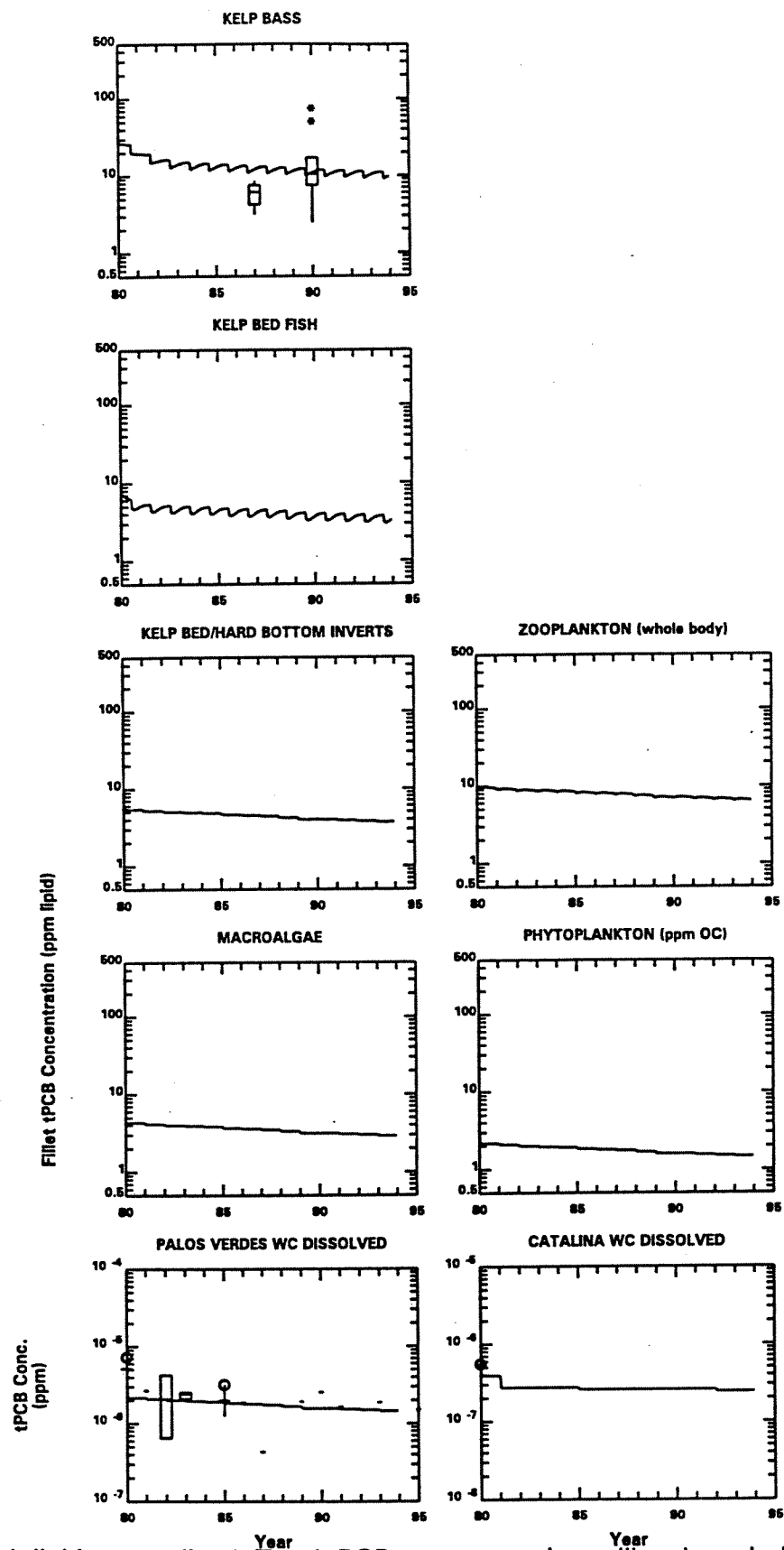
KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

p,p - DDE

KELP BASS

KELP BED FISH

KELP BED/HARD BOTTOM INVERTS

ZOOPLANKTON (whole body)

MACROALGAE

PHYTOPLANKTON (ppm OC)

PALOS VERDES WC DISSOLVED

CATALINA WC DISSOLVED

Fillet ppDDE Concentration (ppm wet)

ppDDE Conc. (ppm)

Year

Year

Figure C3-31. Computed wet weight p,p'DDE concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island).
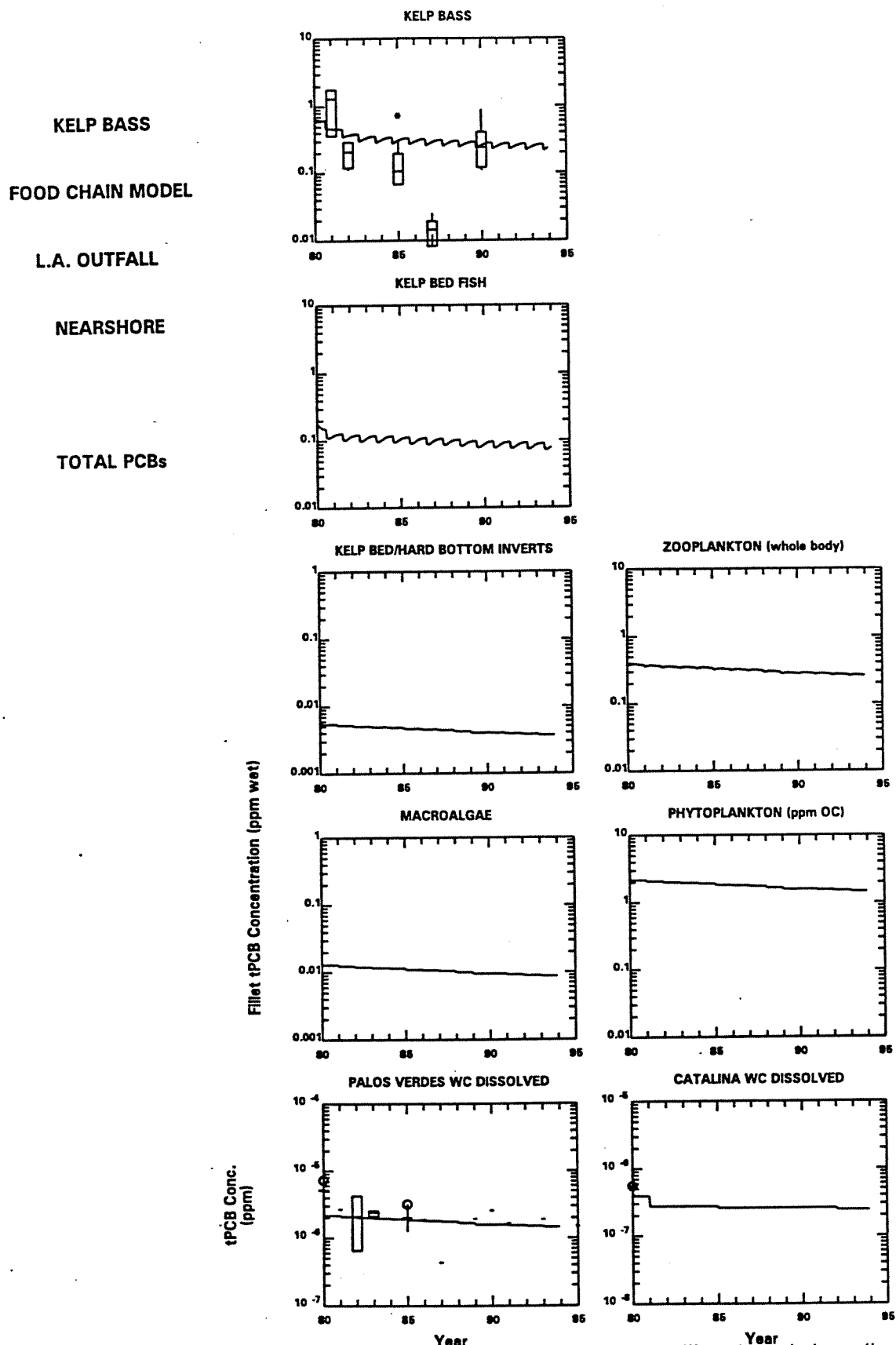
KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

TOTAL PCBs

Figure C3-32. Computed lipid normalized Total PCB concentrations (lines) and data
(boxes) for the kelp bass food web and water column exposure
concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass
migrate to zone of low contamination (Santa Catalina Island).

KELP BASS

FOOD CHAIN MODEL

L.A. OUTFALL

NEARSHORE

TOTAL PCBs

Figure C3-33. Computed wet weight Total PCB concentrations (lines) and data (boxes) for the kelp bass food web and water column exposure concentrations near the LA outfall (Segments 8 and 9). Adult kelp bass migrate to zone of low contamination (Santa Catalina Island).
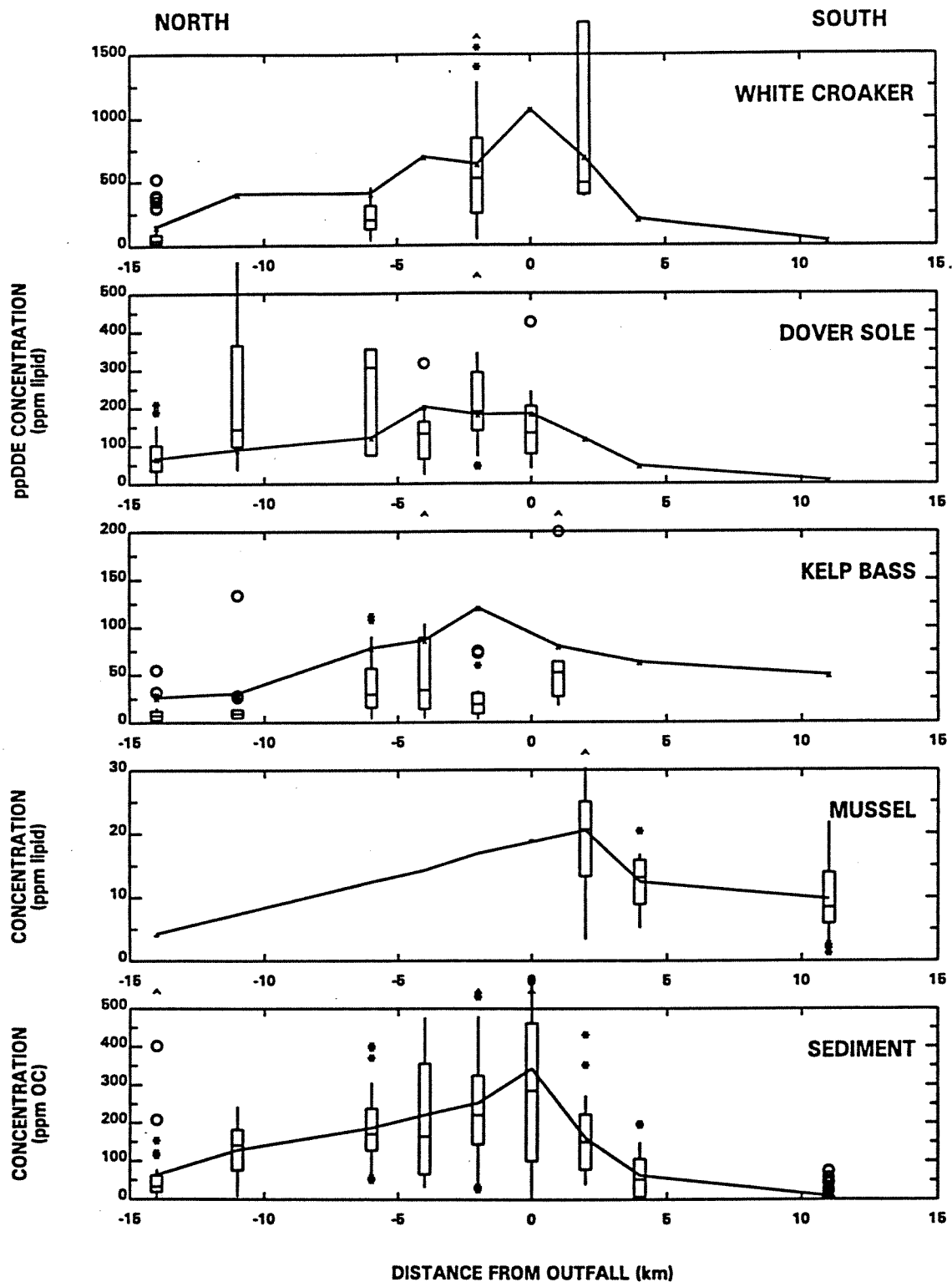
Figure C3-34. Comparison of computed and observed p,p'DDE concentrations for white croaker, Dover sole, kelp bass, mussels and sediments plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). Solid lines indicate steady-state p,p'DDE concentrations predicted by the food web model. Concentrations in fish fillets and mussels are expressed as ppm lipid and surficial sediment concentrations are expressed as ppm organic carbon.
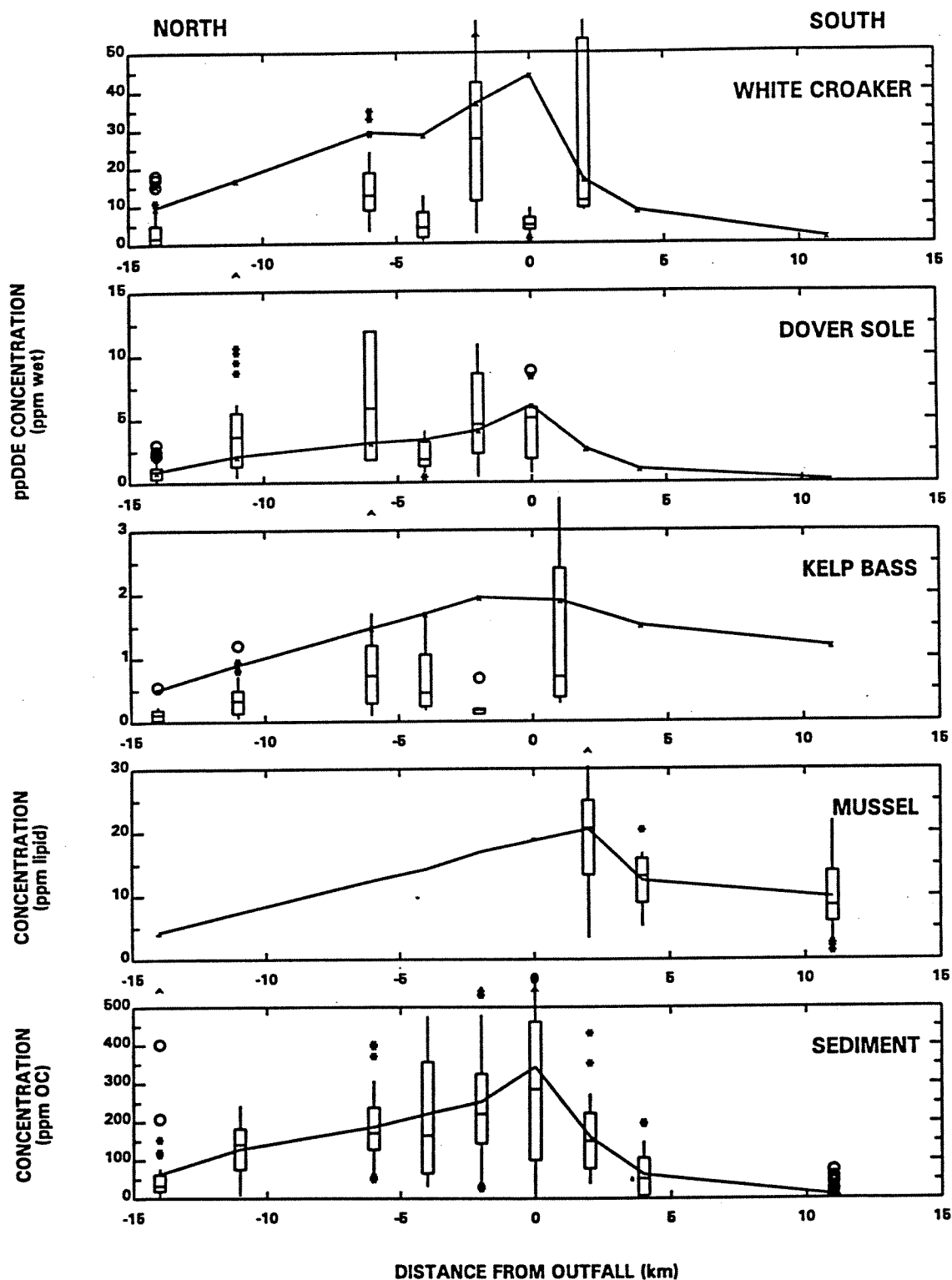
Figure C3-35. Comparison of computed and observed p,p'DDE concentrations for white croaker, Dover sole, kelp bass, mussels and sediments plotted as a function of distance from the Los Angeles County Outfall (kilometer 0). Solid lines indicate steady-state p,p'DDE concentrations predicted by the food web model. Concentrations in fish fillets and mussels are expressed as ppm wet weight and surficial sediment concentrations are expressed as ppm organic carbon.
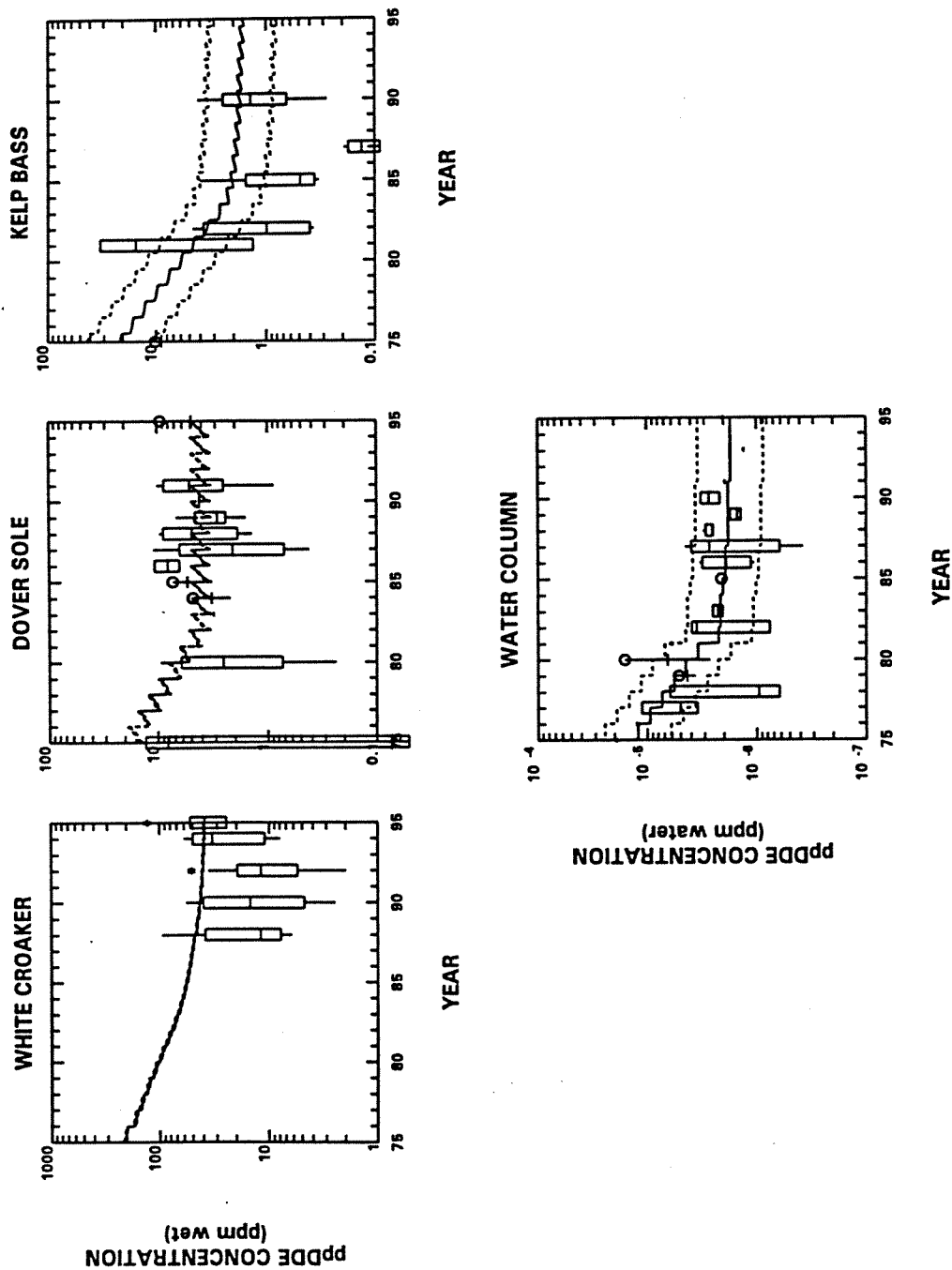
Figure C3-36. Computed wet weight p,p'DDE concentrations (lines) and data (boxes) for white croaker, Dover sole, kelp bass and water column for zone of high contamination. Solid lines indicate model results presented in Sections 3.5 through 3.7; dashed lines indicate computed concentrations when water column concentrations are doubled and halved (ppm wet weight).
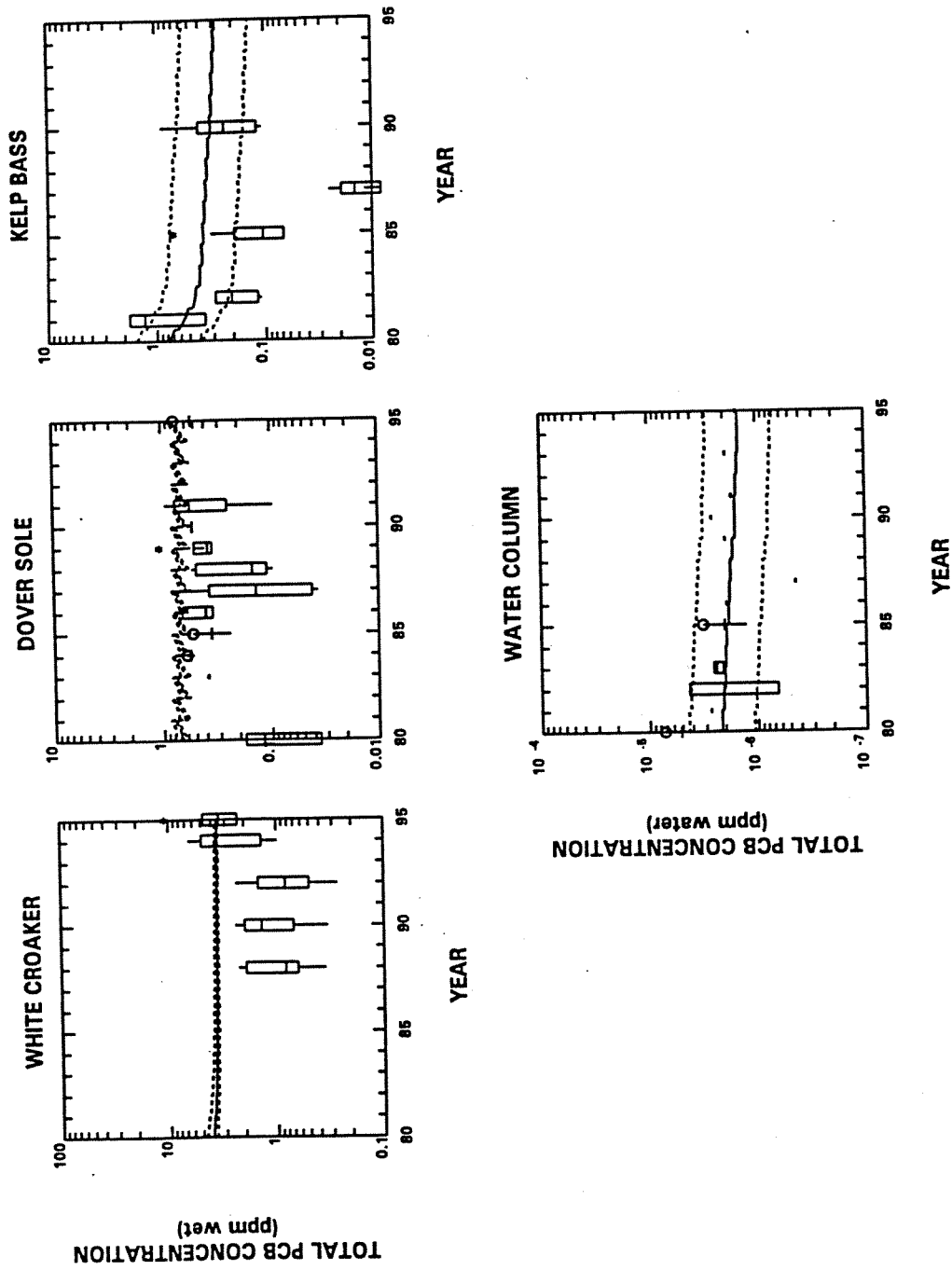
Figure C3-37. Computed wet weight Total PCB concentrations (lines) and data (boxes) for white croaker, Dover sole, kelp bass and water column for zone of high contamination. Solid lines indicate model results presented in Sections 3.5 through 3.7; dashed lines indicate computed concentrations when water column concentrations are doubled and halved (ppm wet weight).

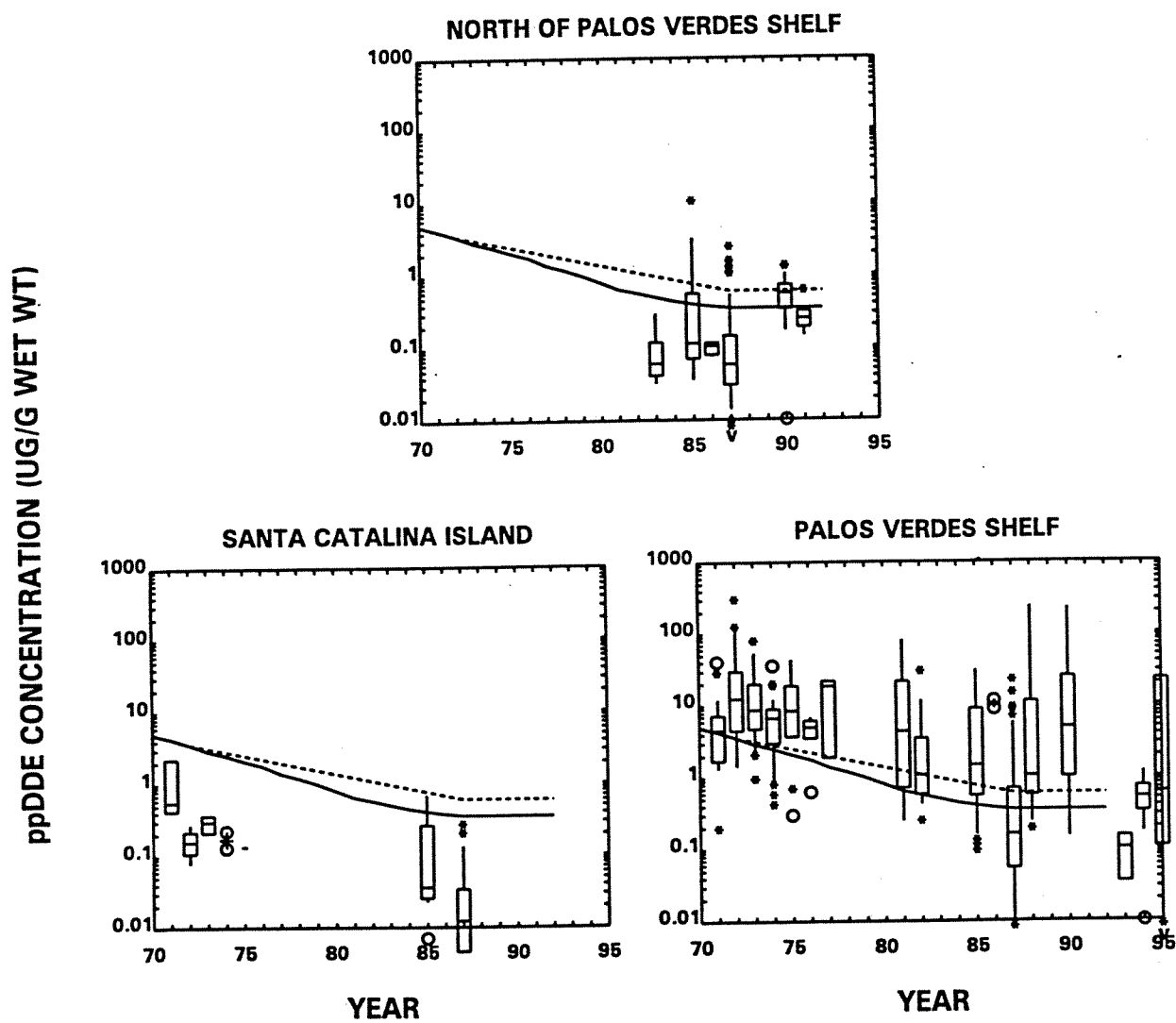# CONTAMINATION IN FISH PREY
OF THE CALIFORNIA SEA LION

Figure C4-12. Comparison of computed prey p,p'DDE concentration profiles (lines) and concentration profiles observed in sea lion prey species for three regions of the Southern California Bight (boxes). Solid lines indicate old GERG data; dashed lines indicate new GERG data.
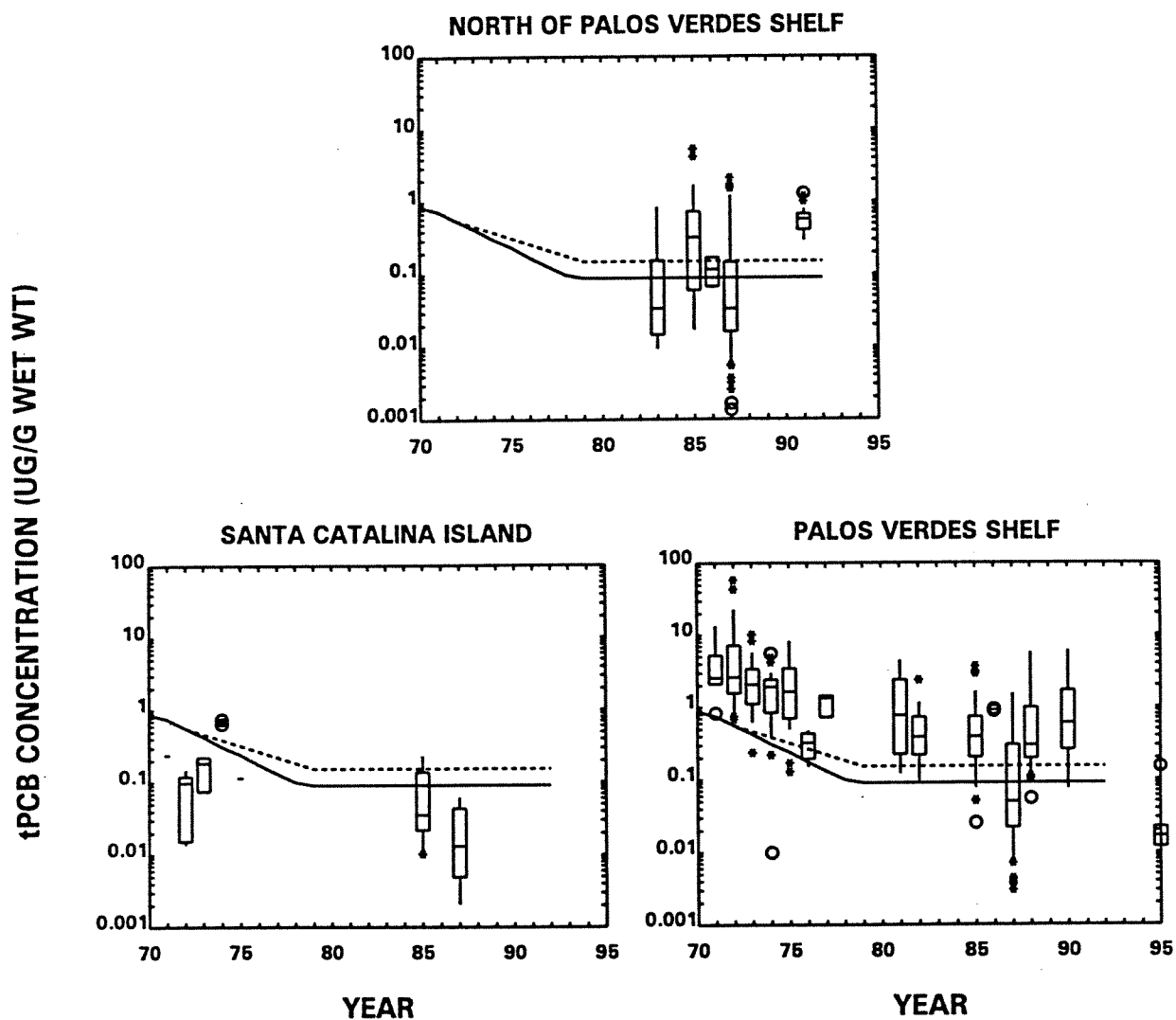
Figure C4-13. Comparison of computed prey PCB concentration profiles (lines) and concentration profiles observed in sea lion prey species for three regions of the Southern California Bight (boxes) Solid lines indicate old GERG data; dashed lines indicate new GERG data.

APPENDIX D

SUMMARY OF HYDROQUAL DATABASE

HydroQual, Inc.
Environmental Engineers and Scientists

Table D2-1. A Summary of the Data in the HydroQual Southern California DDT and PCB Database

Agency	Year	Number of Records		
		Fish	Mussel	Sediment
California Mussel Watch Program	77	-	56	-
	78	-	61	-
	79	-	21	-
	80	-	17	-
	81	-	12	-
	82	-	40	-
	83	-	16	-
	84	-	5	-
	85	-	25	-
	86	-	15	-
	87	-	9	-
	88	-	13	-
	89	-	6	-
	90	-	15	-
	91	-	16	-
	92	-	6	-
	93	-	15	-
	94	-	3	-
	95	-	22	-
Hyperion Treatment Plant	87	6	-	-
	88	93	-	-
	89	117	-	-
	90	94	-	-
LA County Sanitation District	70	3	-	-
	71	192	-	-
	72	298	-	-
	73	155	-	44
	74	205	-	-
	75	110	-	-
	76	44	-	-
	77	67	-	42
	78	2	-	-
	79	1	-	-
	80	19	-	-
	81	24	-	28
	82	81	-	-
	83	24	-	-
	84	8	-	-
	85	128	-	44
	86	15	-	-
	87	24	-	365
	88	103	-	42
	89	21	-	-
	90	129	-	34

Table D2-1. A Summary of the Data in the HydroQual Southern California DDT and PCB Database

Agency	Year	Number of Records		
		Fish	Mussel	Sediment
	91	76	-	-
	92	116	-	34
	93	3	-	-
	94	30	-	14
	95	49	-	-
NOAA Benthic Surveillance	84	-	-	52
	85	-	-	48
	86	-	-	59
	87	-	-	6
	88	-	-	12
	89	-	-	9
	93	-	-	25
	94	-	-	23
	95	-	-	3
NOAA Mussel Watch	84	124	-	-
	85	158	-	-
	86	167	132	218
	87	131	133	180
	88	136	138	3
	89	-	137	36
	90	-	178	27
	91	-	-	27
	94	-	33	-
	95	-	35	-
Pollock et al. (1991)	87	996	-	-
Santa Monica Bay Restoration Project	90	71	-	-
Southern California Bight Damage Assessment	91	-	-	213
	92	146	-	-
Southern California Coastal Water Research Project Pilot Study	94	-	-	219
	95	-	-	74
Garcelon et al. (1994a,b)	92	29		
	93	3		
Martin et al. (1984)	83	-	6	-
Young (1982)	74	-	2	-

Table D2-1. A Summary of the Data in the HydroQual Southern California DDT and PCB Database

Agency	Year	Number of Records		
		Fish	Mussel	Sediment
Young et al. (1982)	71	-	1	-
	72	-	1	-
	73	-	1	-
	74	-	1	-
	75	-	1	-
	76	-	1	-
	77	-	1	-
	78	-	1	-
	79	-	1	-
	80	-	1	-
	81	-	1	-
Young and Heesen (1978)	71	-	17	-
Young et al. (1978)	73	9	-	-
	74	8	-	-
	75	161	-	-
	76	161	-	-
	77	18	-	-
	79	1	-	-
Risebrough (1987)	71	-	9	-
	72	-	2	-
	73	-	2	-
	74	-	4	-
	77	-	6	-
	78	-	6	-
	85	131	13	21
	86	-	2	-
Overall		4687	1239	1902

APPENDIX E

PROBABILITY PLOTS FOR WHITE CROAKER, KELP BASS, AND DOVER SOLE

APPENDIX E

PROBABILITY PLOTS FOR WHITE CROAKER, KELP BASS, AND DOVER SOLE

This appendix includes probability plots for white croaker, kelp bass, and Dover sole for each HydroQual segment for the analysis presented in Chapter 6. Plots are presented for all tissues and associated key values listed in Table 6-1. The segment number is posted in the upper left hand corner of each panel, and the number of observations for each segment is posted in the lower right hand corner.

Muscle concentrations (ppm wet weight).
Total DDT for white croaker, 1985 to 1995.

Muscle concentrations (ppm wet weight).
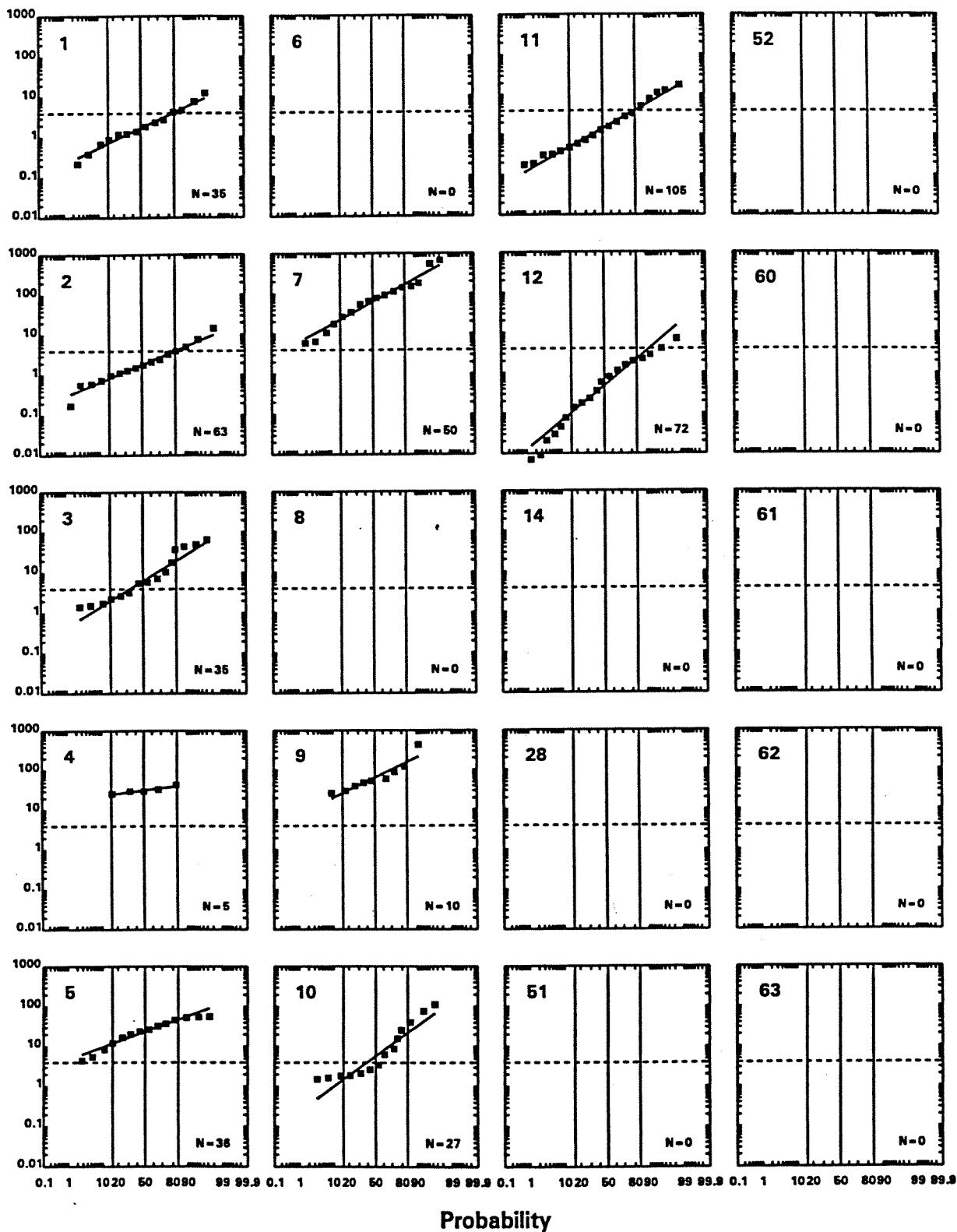Total DDT for kelp bass, 1985 to 1995.

Muscle concentrations (ppm wet weight).
Total DDT for dover sole, 1985 to 1995.
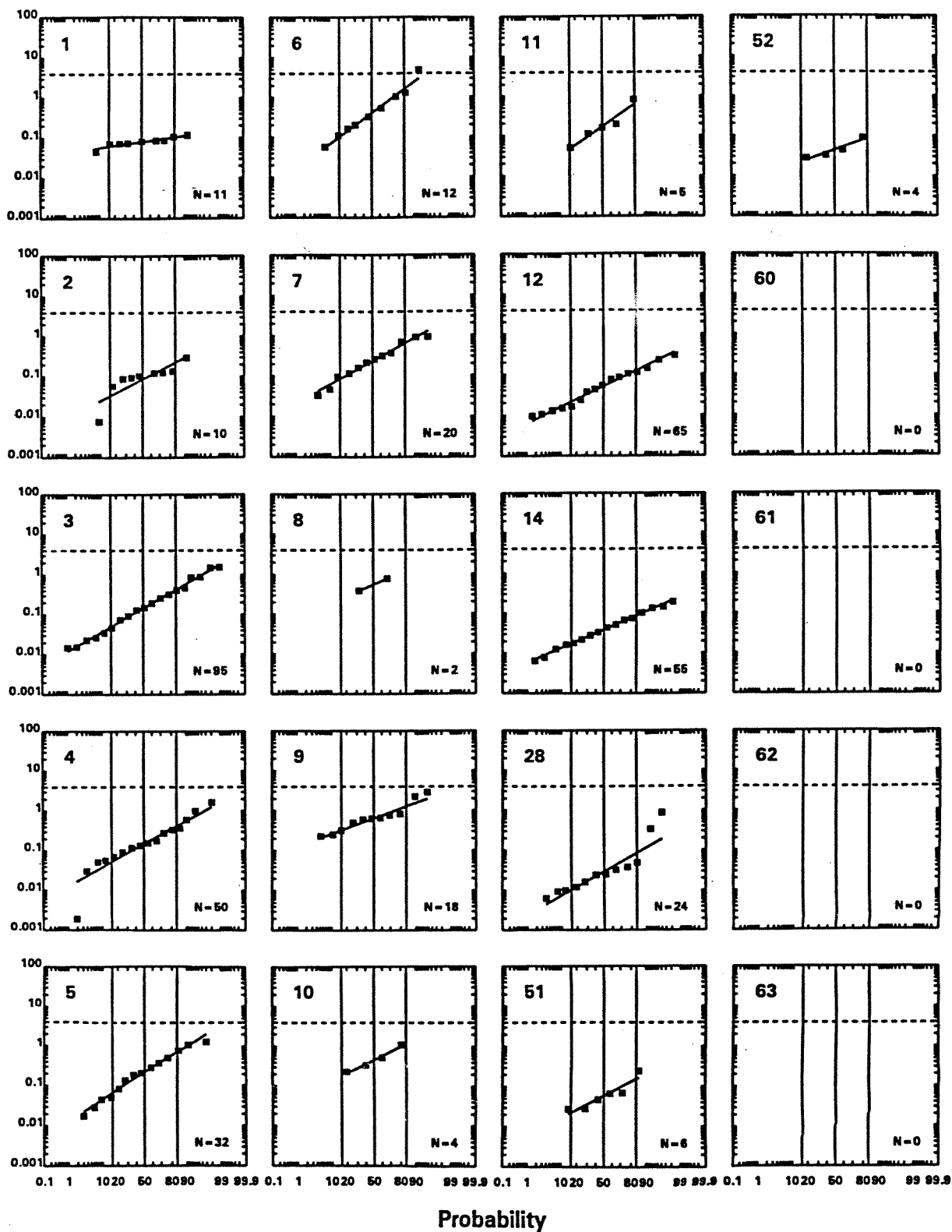
Total PCB concentration in muscle tissue (ppm wet weight)

Muscle concentrations (ppm wet weight).
Total PCB for white croaker, 1985 to 1995.
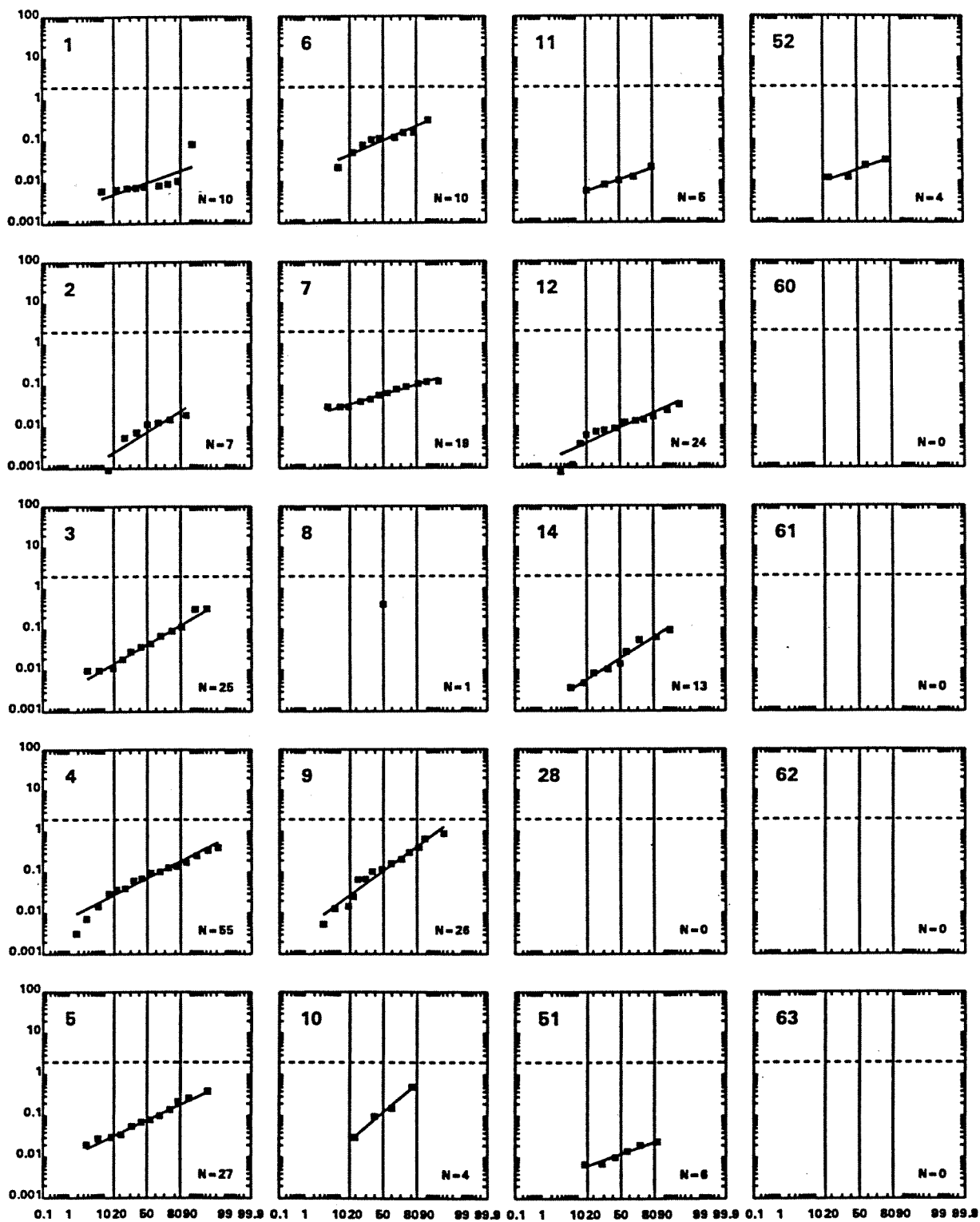
Muscle concentrations (ppm wet weight).
Total PCB for kelp bass, 1985 to 1995.
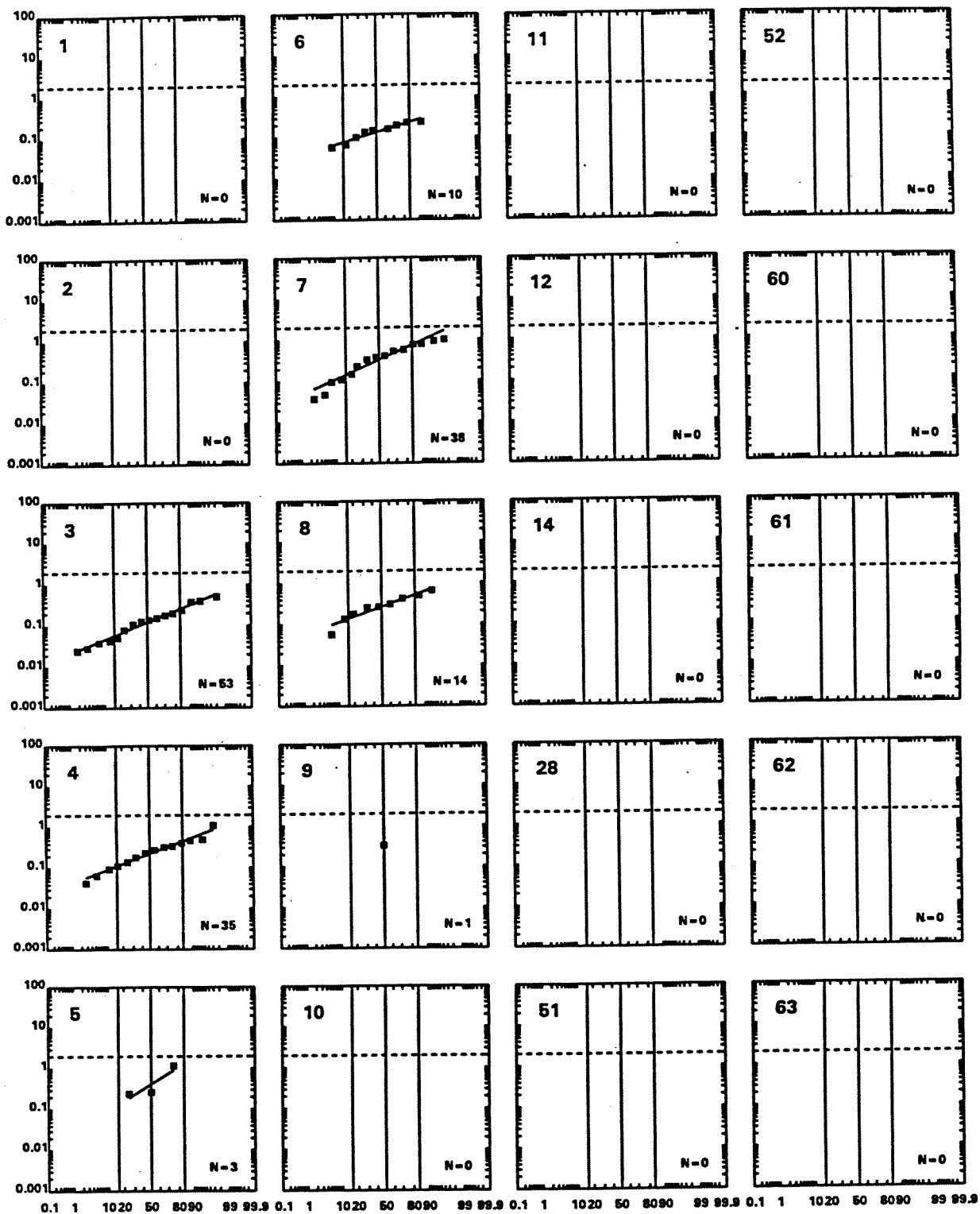
Muscle concentrations (ppm wet weight).
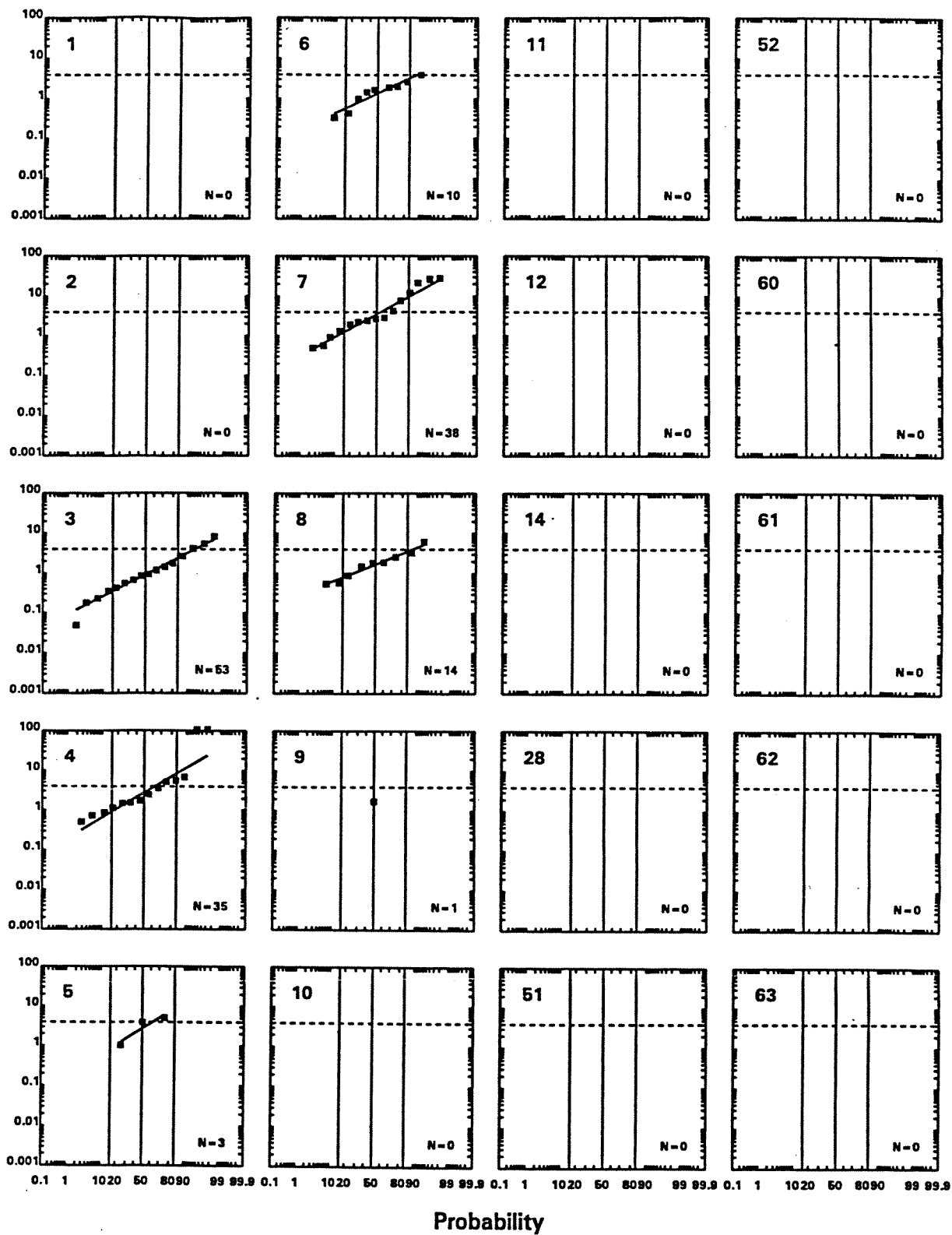Total PCB for dover sole, 1985 to 1995.

Ovary and ovary equivalent concentrations (ppm wet weight).
Total DDT for white croaker, 1985 to 1995.

Ovary equivalent concentrations, (ppm wet weight).
Total DDT for kelp bass, 1985 to 1995.

Ovary equivalent concentrations, (ppm wet weight).
Total DDT for dover sole, 1985 to 1995.